

Glacier National Park Fisheries and Aquatic Invasive Species Programs 2018-2019 Annual Report



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Front cover photo caption: Adult lake trout suppression in Quartz Lake, fall 2018.

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Quartz Lake Lake Trout Suppression Project

ABSTRACT

Approximately one-third of the USFWS-designated natural lake core areas supporting threatened bull trout *Salvelinus confluentus* are found in Glacier National Park (GNP). However, bull trout populations on the west side of GNP have declined in recent decades and are at high risk of extirpation in several lakes due to the establishment of invasive lake trout. In 2009, the U.S. Geological Survey and the National Park Service began suppressing lake trout in Quartz Lake (352 ha) to reduce impacts to bull trout and other native fishes. In 2019, we removed 279 lake trout. A total of 9,848 lake trout have been removed since the project began in 2009. Evident in the length-frequency data is an absence of new juvenile recruits to our smallest mesh sizes. Our annual catch of juvenile lake trout continues to decline rapidly, with the 2019 catch substantially smaller than the 2018 catch. This could signal the beginning of an impact to juvenile production from the adult removal efforts. Catch of adult lake trout has been increasing over the past few years. There are clearly more adult sized lake trout in Quartz Lake than we have seen for a number of years. Both angler catch rates, gill netting CPUE, and adult population estimates all confirm this. The increase in adult lake trout is likely the result of successful reproduction and recruitment early in our suppression efforts when we were still experimenting with different techniques. However, based on the limited reproductive success indicated by our recent juvenile lake trout catch's, we expect adult numbers to drop off dramatically in the next 2-4 years as the weak year classes of juvenile lake trout mature. As we continue to remove the adults at a high rate, the cycle should feed on itself and result in fewer and fewer lake trout.). Native fish numbers, as evidenced by stable angler catch rates and bull trout redd counts, remain stable. This is the primary goal of the project. Additional years of netting will be required to better clarify trends in the lake trout population as well as the overall effectiveness of our strategy of removing both adult and juvenile lake trout annually using targeted gill netting to collapse the population.

INTRODUCTION

Aquatic invasive species threaten the biodiversity of native aquatic ecosystems worldwide and are considered the one of greatest threats to native biodiversity in North America (Vitousek et al. 1997). Invasions of introduced taxa often disrupt the structure and function of ecosystems, reduce biological diversity among native species, and impose huge economic costs. Therefore, understanding how to reduce or eliminate invasive species in native ecosystems is critical for conserving native aquatic species and ecosystems.

Lake trout are large, long-lived, top-level predators native to deep, cold, oligotrophic lakes of Canada and northern parts of the United States, including the Great Lakes (Behnke 2002). During the late 19th and early 20th century, lake trout were widely introduced into lakes and reservoirs outside their native range (Crossman 1995). More recently, the species has expanded its range in the western United States through dispersal and unauthorized translocations (Behnke 2002). Bull trout and lake trout have similar morphologies, diets, and growth rates, and occupy similar trophic positions as top-level predators, resulting in strong potential for competitive or predatory interactions. In the Pacific Northwest, native bull trout populations have generally declined in systems where lake trout have been introduced (Donald and Alger 1993; Fredenberg 2002; Martinez et al. 2009).

Introduced to Flathead Lake in 1905, non-native lake trout have dispersed to many additional lakes in the wider Flathead watershed, including nine lakes west of the Continental Divide in GNP that historically supported bull trout (Fredenberg 2002; Downs et al. 2011). Fredenberg (2002) reported a broad bull trout decline and a corresponding increase in non-native lake trout from 1969 to 2000 in four lakes (Logging, Bowman, Harrison, Kintla) in GNP. Recent surveys produced similar results (Figures 1-3). Bull trout have continued to comprise a similar proportion of standardized gill net catch from 1969-2009 (Downs et al. 2011), presumably due to the absence of lake trout (Figure 4). Standardized gill net monitoring on Quartz Lake was halted in 2010 because lake trout gill netting removal began in 2009 and there was a desire to eliminate additional mortality impacts on the bull trout population during these efforts. Similarly, recent bull trout redd surveys (2003-2019) show continued decline in adult abundance in many GNP waters, with some counts representing less than 10 reproducing individuals. Combined, these data show that several native bull trout populations have drastically declined in western GNP with several adfluvial populations at imminent risk of extirpation due to adverse interactions with invasive lake trout.

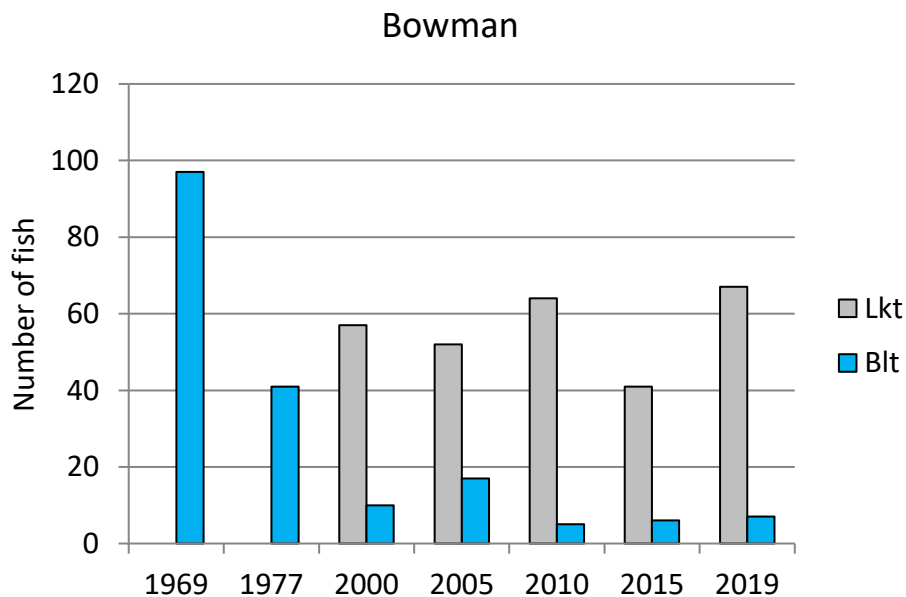


Figure 1. Bowman Lake trend gill netting results over time.

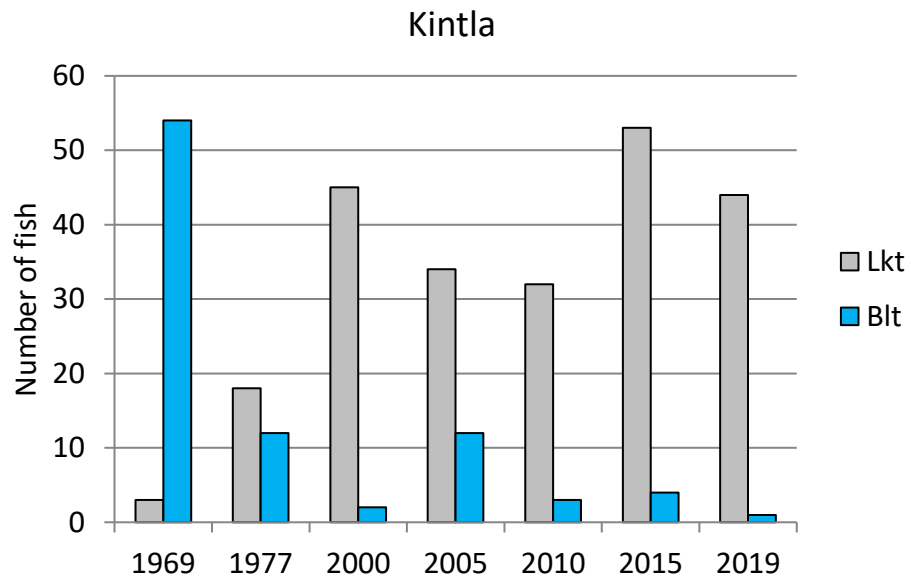


Figure 2. Kintla Lake trend gill netting results over time.

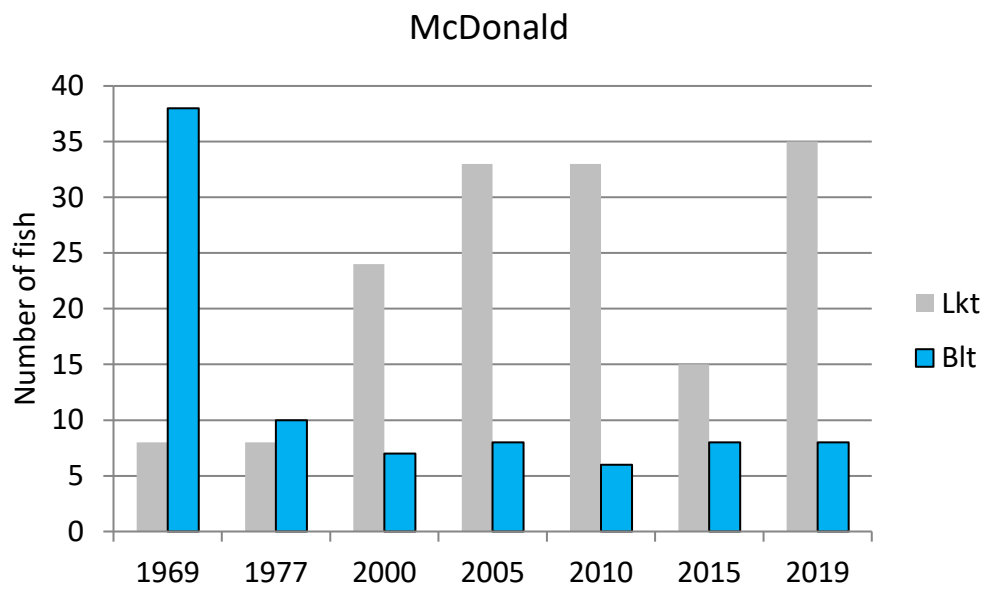


Figure 3. Lake McDonald trend gill netting results over time.

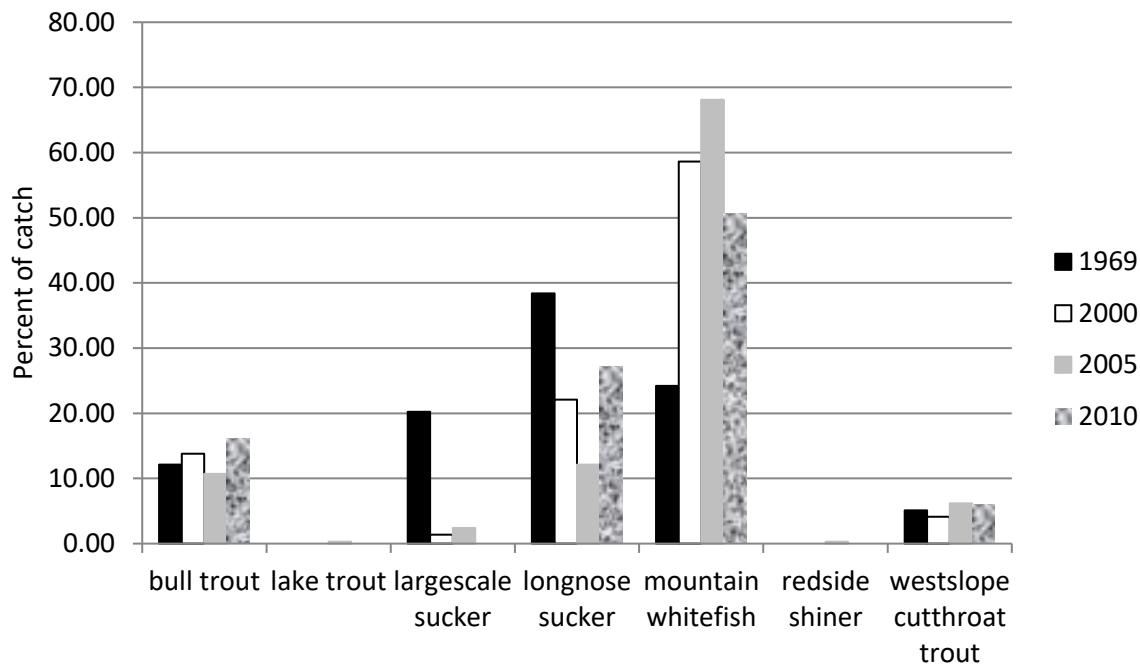


Figure 4. Quartz Lake trend gill netting results over time.

STUDY AREA

Quartz Lake is a glacially formed lake located in the headwaters of the Columbia River Basin, Montana. Quartz Lake is the fifth largest lake west of the Continental Divide in GNP (Fredenberg et al. 2007), with a surface area of 352 ha and a maximum depth of 83m (Figure 5). The lake is at an elevation of 1,346 m and is positioned in a narrow-glaciated valley that is supplied by perennial flow from snow and glacial runoff from the Lewis Range. The limnetic zone substrate is dominated by a mixture of cobble and boulder. Quartz is an oligotrophic, dimictic lake with stratification occurring in late June and destratification in early October. Three avalanche chutes along the northern shoreline have deposited large rock substrates into the lake that are suitable for lake trout spawning.

The native fish assemblage in the Quartz drainage consists of bull trout, westslope cutthroat trout *Oncorhynchus clarkii lewisi*, mountain whitefish *Prosopium williamsoni*, longnose sucker *Catostomus catostomus*, largescale sucker *Catostomus macrocheilus*, slimy sculpin *Cottus cognatus*, and redbside shiner *Richardsonius balteatus*. Lake trout is the only non-native fish species in the drainage (there is also likely some hybridization between rainbow and westslope cutthroat trout occurring in Quartz Creek downstream of Quartz Lake) and was first detected in Lower Quartz Lake in 2003 and in Quartz Lake in 2005 (Fredenberg et al. 2007).

There are no natural fish barriers in the Quartz drainage, although several high gradient cascades serve as potential intermittent barriers between Lower Quartz and Middle Quartz lakes. The discovery of lake trout in Lower Quartz Lake prompted construction of an artificial gabion barrier approximately 100 m downstream of Middle Quartz Lake in 2004 to conserve the upstream native fish assemblages in Middle Quartz, Quartz, and Cerulean lakes. Unfortunately, lake trout were detected in Quartz Lake in 2005 before the barrier was completed. The long-term functionality of a barrier remained important to the success of lake trout suppression in Quartz Lake and consequently the NPS removed the existing structure which had fallen into disrepair and constructed a new barrier structure in 2012 to deter further invasions.

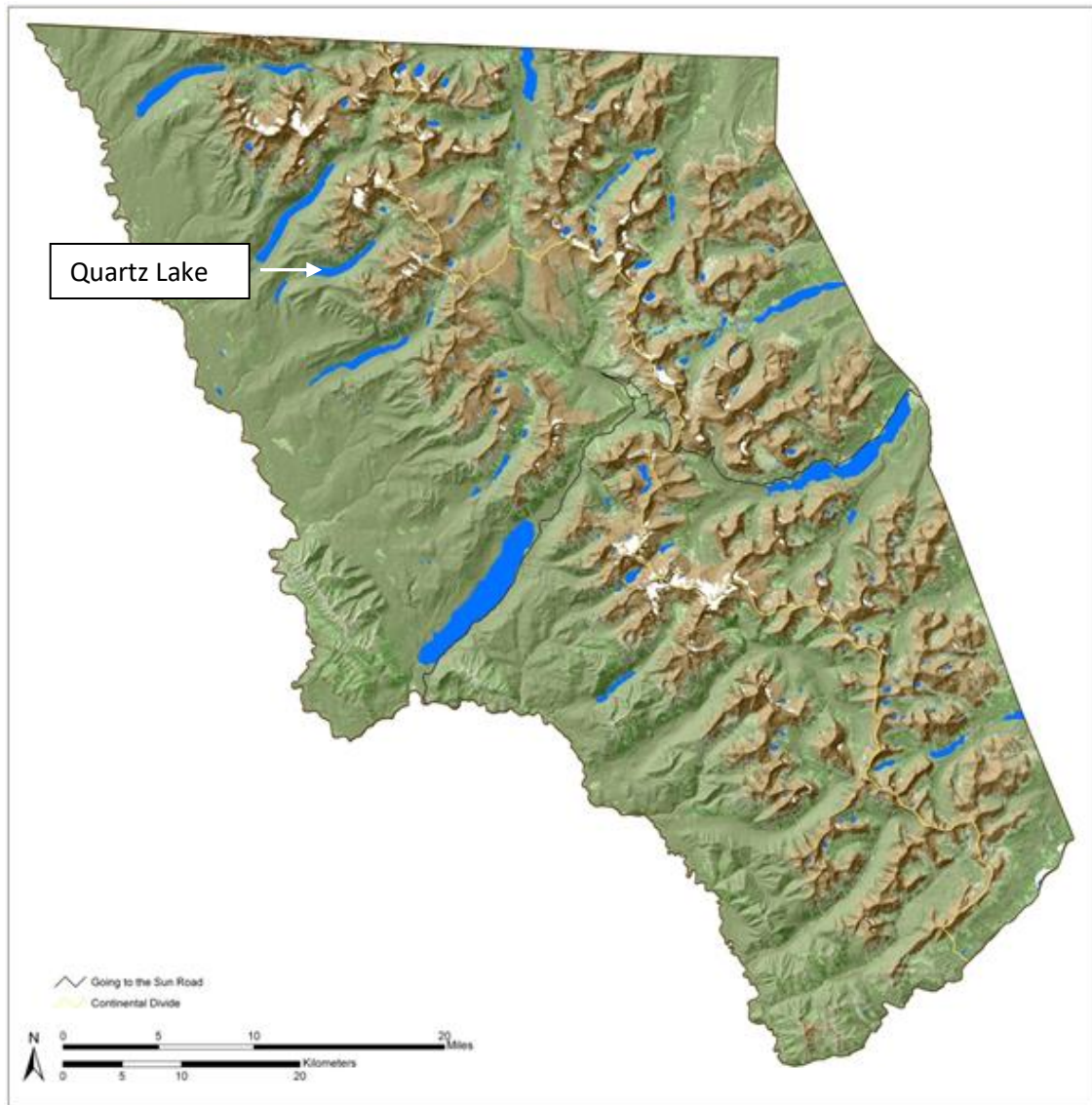


Figure 5. Location of Quartz Lake in the North Fork of the Flathead River Drainage, Glacier National Park.

METHODS

Telemetry

In 2019, we used sonic telemetry to track movements of adult lake trout to spawning areas and to conduct a mark and recapture population estimate. Nine adult-sized lake trout were captured by anglers in June and were marked with sonic tags. Fish were anesthetized with MS-222 and a sonic tag was implanted into the abdominal cavity of the fish. The surgical wound was closed with monofilament Sucher material. Each adult tag was 63mm long, 15.6 mm in diameter, weighed 10g in water, and lasted 36 months. Ten juvenile lake trout (<325mm;TL) were captured in gill nets and also marked with small sonic tags. Each tag was 23mm long, 9mm in diameter, weighed 2.3g and lasted 90 days. Juveniles were tracked throughout the month of June while adult lake trout were tracked in September and October.

Gillnetting

We recorded length (mm; Total Length (TL) and weight (g) from all lake trout captured. Otoliths and genetic samples were also taken for future aging and movement studies. Additionally, otoliths, genetic samples, and weights were taken from all bull trout and westslope cutthroat trout bycatch mortalities. All lake trout were killed and their air bladders were punctured before they were returned to the lake.

Spring

The 2019 lake trout suppression program followed the same general schedule utilized in previous years. Gillnetting was comprised of two annual netting periods; spring (31 May-28 June) and fall (4 October-31 October). During spring gillnetting, crews netted from Monday-Friday for a five-week period. Juvenile and subadult lake trout were captured using small mesh (19 mm, 26 mm, and 32 mm bar measure) 91 m-long sinking monofilament gillnets targeting. Nets were set at all hours of the day and were generally deployed for 4-6 hours in depths greater than 40m. The short deployment interval and deeper net sets were purposefully used to reduce incidental bycatch mortality of bull trout and westslope cutthroat trout. In previous years 28 mm bar mesh nets were used, however, selectivity analysis in 2013 revealed the peak length selected by the mesh sizes used overlapped with the other mesh sizes being used, therefore the use of 28 mm bar measure mesh was dropped beginning in 2014. In 2013, catch rates of short (33 m) experimental 19 mm bar measure gillnets with the standard twine diameter of 0.20 mm were compared to catch rates of 19 mm bar measure gillnets containing finer 0.10 mm twine diameter (Fredenberg 2014). It was determined the finer 0.10 mm twine diameter had a significantly higher catch per unit effort than the 0.20 mm twine diameter (Fredenberg 2014) and finer mesh nets were used exclusively starting in 2014. However, the durability of the 0.10 mm nets was problematic and the cost of replacing nets would be unreasonable, therefore, beginning in 2015, full length (91 m) nets of 19 mm, 26 mm and 32 mm bar measure mesh and 0.15 mm twine diameters were used in the spring. The spring objective is to maximize removal of juvenile lake trout as young as possible in order to reduce recruitment to the adult population.

Fall

The second netting event (fall netting) occurs throughout October and targets mature adult lake trout, primarily on the two known spawning areas as determined by earlier telemetry efforts (2009-2014) and the utilization of the locations of adult lake trout that were tagged in 2017 and 2019. The fall netting objective is to remove as many adults as possible in order to reduce the number of juveniles produced in subsequent years. Adult lake trout are captured with large mesh (57 mm, 64 mm, and 70 mm bar measure), 91 m-long sinking monofilament gillnets set in relatively shallow (<10m) areas of known spawning sites. Once completely set, these nets were allowed to fish for approximately one hour before we started to pick up the end of the net that was set first. This provided for a 1.5-2-hour soak time for all portions of the net. Short set times were used to minimize bycatch mortality of native fish, primarily westslope cutthroat trout and bull trout. We generally make a total of 4-6 sets/night.

RESULTS AND DISCUSSION

Telemetry

As the suppression years have progressed from 2010-2019, anglers have found it increasingly difficult to capture mature size lake trout for sonic tagging. We caught 9 adult lake trout >450 mm via hook and line sampling and surgically implanted the sonic transmitters in 2019. In addition, we continued to track an at-large adult lake trout that was tagged in 2017. Four of these fish were recaptured in nets during fall netting efforts. Avalanche chutes 1-3 are the locations that were previously identified as primary bull trout spawning sites (Fredenberg 2014), with chutes 2 and 3 being the most used. However, telemetry data from 2019 suggests additional spawning areas may be located on the south bank across from chute 1. There were no extreme congregations of numerous tagged fish and only a handful of nights when

multiple tagged fish were located in close proximity of one another. Previous telemetry data indicates tagged adult lake trout used all areas of the lake throughout the month of October and most of the adults we tagged also moved to at least one of three known spawning locations at some point (Figures A.1 to A.10). This suggests that we are being successful in using our current technique of netting the known spawning locations, but we need to also net some of the other potential spawning sites identified in the most recent telemetry.

We recaptured 4 of the 9 adults that were radio-tagged in 2019, resulting in a population estimate of spawning adults of 143 (95% CI: 62-360). Adult abundance in 2019 is similar to the adult abundance estimate of 146 for 2009, the first year of the project (Fredenberg 2014) (Figure 6). When compared to other years, this is a low recapture rate. We generally get 80-90% recapture of our tagged fish. In 2019, that number fell to 44%. That means approximately half of the lake trout in Quartz Lake were able to spawn and we will have to increase our juvenile and adult netting in the future to mitigate the impacts on lake trout population growth. Three of the nine tagged adults we did not recapture were all located in approximately 20-35m of water, directly off of the public campground in late October. Two were only re-located in this area and it is difficult to tell if they were mortalities, a spawning congregation, or for some other reason. This area should be netted using the adult nets in 2020 to better understand the use of the area by adult lake trout.

Eleven juvenile lake trout (<300mm) were tagged with sonic transmitters and released in 2019. Only seven of these were subsequently relocated in the lake (Figures A.11-A.17). The intent of the tagging was to determine if juvenile lake trout alter their movement behavior as June progresses, resulting in reduced vulnerability to our nets over time or if certain fish have a preference for deep or shallow water. In addition, recovery of these fish in our spring sampling nets would offer the opportunity to conduct a mark-recapture population estimate for juvenile lake trout, however none of the tagged juveniles were recaptured in our nets in 2019. Five of seven juvenile lake trout occupied a wide range of depths during tracking, from shallow areas (<10m) to the deepest portions of the lake (40-70+m). One of the tagged juvenile lake trout (tag 8) did not move a great deal post-surgery and is presumed to have been a mortality. The other fish that we did not document inhabiting deep water (tag 6) was only relocated twice but had travelled from one end of the lake to the other. This fish could have occupied deeper depths that we did not document due to the sparse relocations. In addition, most fish moved throughout the lake over the spring. These data suggest that despite concentrating our netting efforts in waters >40m in depth to reduce bycatch and not netting the head of the lake, most if not all juvenile lake trout would be vulnerable to our nets at some point.

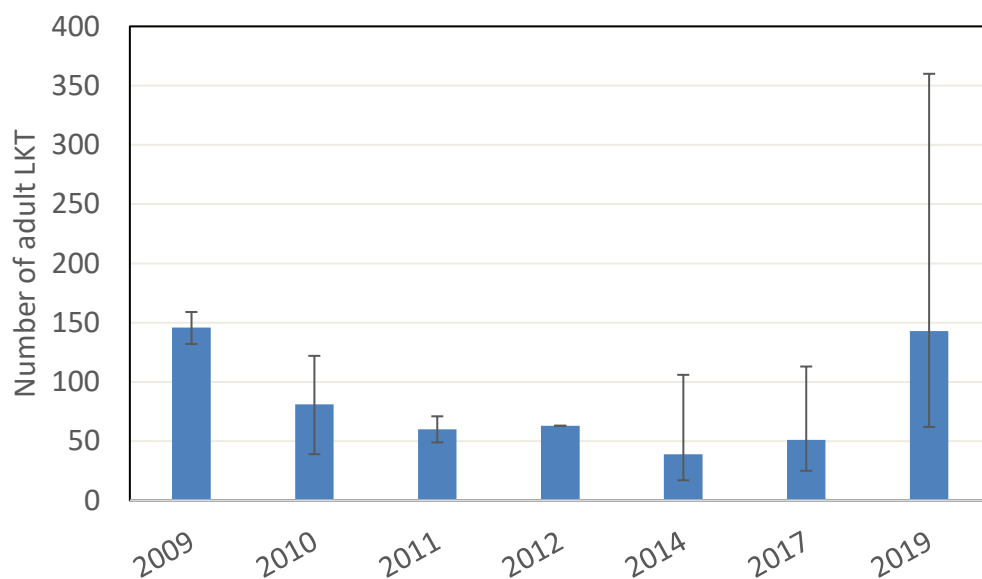


Figure 6. Mark and recapture abundance estimates for adult lake trout in Quartz Lake, 2009-present.

While lake trout catch per angling hour declined substantially between 2010 and 2015, the relationship increased from 2016 to 2018 and held steady in 2019 (Figure 7). This is likely a reflection of more catchable sized lake trout recruiting to our gear. These fish were likely produced early in the programs implementation when netting was likely less efficient. In addition, it could be partially the result of our increased knowledge in how to effectively target lake trout in Quartz Lake with angling gear. It could take several more years of netting to determine if the mechanism and if netting can overcome this increase. The bull trout catch per angler hour remained relatively stable from 2010 to 2014 but has fluctuated for the past three years (Figure 7). These downward fluctuations may be a reflection of anglers becoming better at targeting lake trout without incidentally catching bull trout.

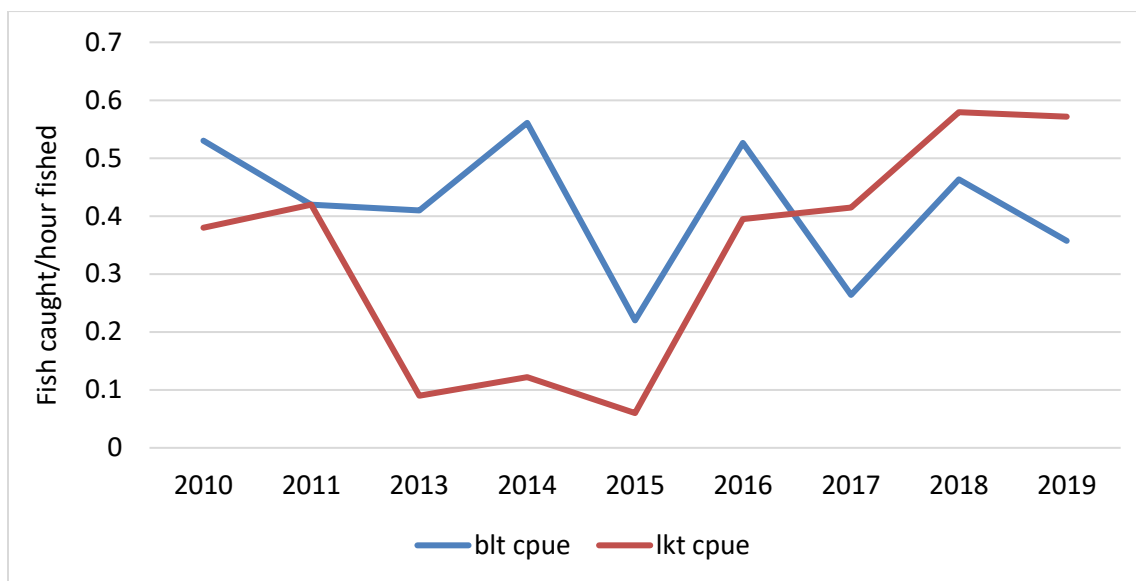


Figure 7. Catch-per-unit-effort (fish/angler hour) for project personnel from 2010 - 2019.

Gillnetting

Spring

Approximately 42km of net was deployed in 2019, including 29.4km during the spring and 12.6 km during the fall, resulting in a total of 279 lake trout being removed (204 juveniles and subadults, 75 adults) (Table 1). This brings the total number of lake trout removed from the system to 9,848. We typically spend between 5-6 weeks each spring netting, however in 2019 we only fished about four weeks. It took longer than usual to get the netting operation up and running for the 2019 spring season. The total amount of net set each spring increased dramatically when we put the new gill netting boat into operation in 2016 and we are likely now limited in the amount of net we can set each spring by funding for staff to run the boat on additional days. The program has continually used information gained in earlier years of the program in an adaptive fashion to more efficiently remove juveniles during spring netting while minimizing bycatch of other native fish. The use of fine diameter twine (0.15mm) with a green tint to construct nets has greatly improved our catch of juvenile lake trout. Unfortunately, it also increased our catch of native juvenile whitefish which also occupy a wide range of depth in the lake. The zone of overlap with whitefish appears to occur when we fish our nets in less than 40m of water. When we fish deeper than 40m, we typically catch only lake trout and a few native suckers. The suckers are much “hardier” than the mountain whitefish and readily swim back down when released from the nets. The mountain whitefish have more delicate features and are often injured, killed, and/or de-scaled when being removed from the nets. Post-release survival of mountain whitefish is likely quite low. By only fishing our spring nets in water >40m deep, we are limiting our ability to catch juvenile lake trout as our radio-telemetry indicates they are found at shallower depths. This is one of the trade-offs we make to ensure we are minimizing the harm to native species.

Table 1. Annual kilometers of net set during spring and fall gillnetting from 2009-2019 in Quartz Lake, Montana.

Year	Spring	Fall	Total
2009	--	38	38
2010	23.3	27.3	50.6
2011	28.1	12.8	40.9
2012	18.7	10.2	28.9
2013	17.6	14.0	31.6
2014	17.8	17.3	35.1
2015	16.9	11.9	28.8
2016	36.2	13.0	49.2
2017	23.7	7.9	31.6
2018	48.8	17.6	66.4
2019	29.4	12.6	42.0

Spring netting in 2019 removed a total of 204 juvenile lake trout. An overall declining trend in catch rate of juvenile lake trout from 2013 to present is apparent, suggesting we have dramatically reduced the abundance of juvenile lake trout over time (Figures 8 and 9). Juvenile length at capture in standardized nets (19, 26, and 32 mm bar measure) increased steadily from 2013-2018, before regressing a bit in 2019 (Table 2). This is a result of a lack of recruitment of the smallest sizes of lake trout into our smallest meshes, suggesting we have significantly reduced lake trout reproductive success through adult removal. We did see a slight increase in the number of young lake trout in the smallest size classes in 2019, suggesting variable (but very low) reproductive success in the past few years (Figure 9). Mirroring this trend is an evaluation of catch rate by mesh size (Figure 10), which shows a declining catch trend for the smallest fish sizes (until 2019) with the larger meshes catching a greater proportion of the fish in more recent years. As gill nets are highly size-selective (i.e. small meshes catch small fish while big meshes catch bigger fish), it is clear we have impacted juvenile lake trout production in the lake.

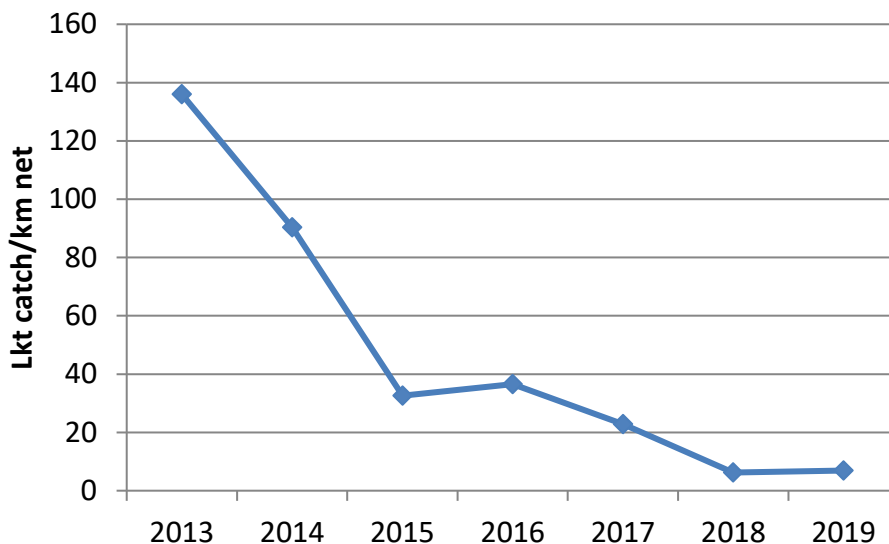


Figure 8. Catch-per-unit-effort (lake trout captured per km of net set) during spring juvenile lake trout removal efforts in Quartz Lake.

Table 2. Mean length over time of juvenile lake trout removed in spring suppression efforts from Quartz Lake.

	2013	2014	2015	2016	2017	2018	2019
Low 95%	228.6	221.4	241.2	270.7	297.8	296.2	250.5
Mean	229.8	222.8	243.3	272.1	300.5	302.3	258.4
Up 95%	231.1	224.2	245.5	273.4	303.3	308.5	266.3

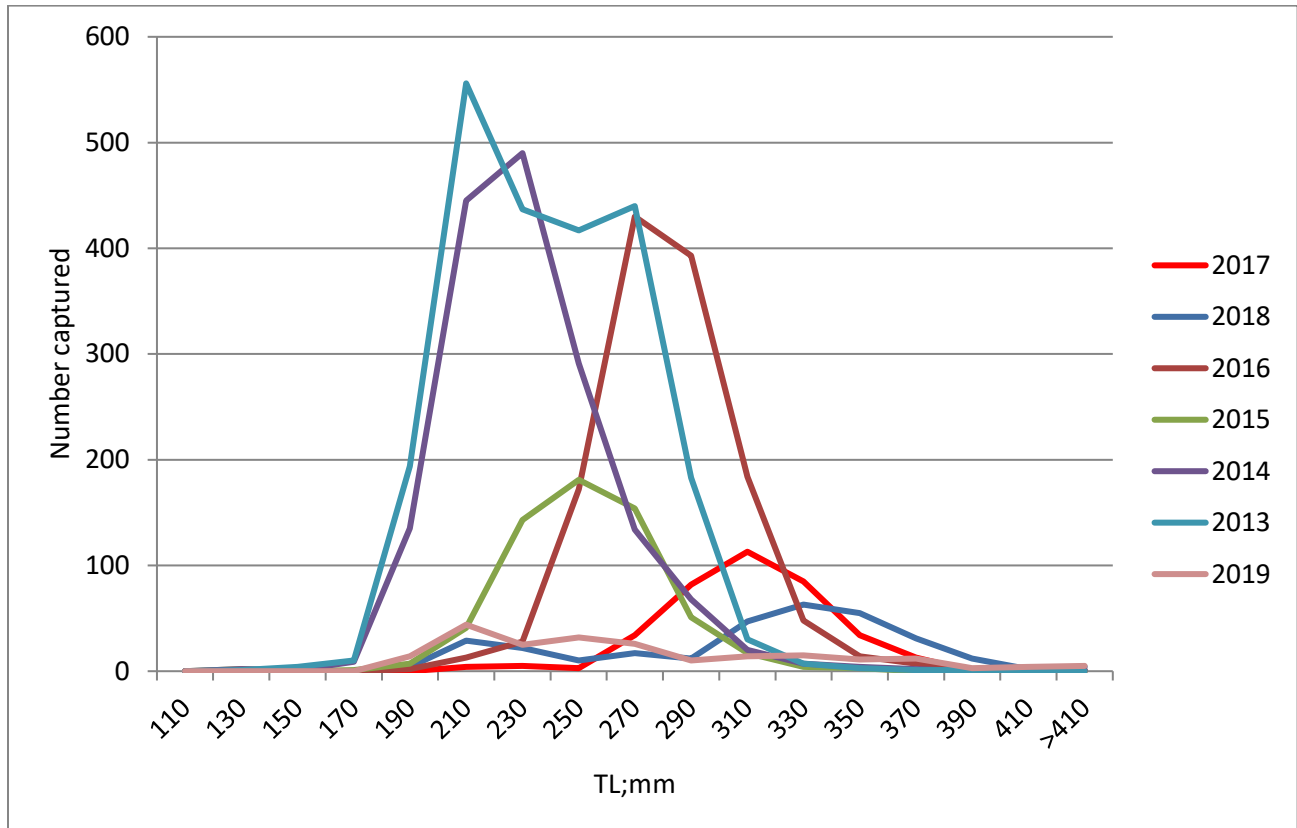


Figure 9. Length frequency histogram of juvenile lake trout captured in standardized gill nets over time (19mm, 26mm, 32mm bar measure mesh and 0.15" twine diameter) during spring in Quartz Lake, Montana. The only difference in net mesh size was the use of 28mm mesh in addition to the other three mesh sizes in 2013. The 28mm mesh was dropped beginning in 2014.

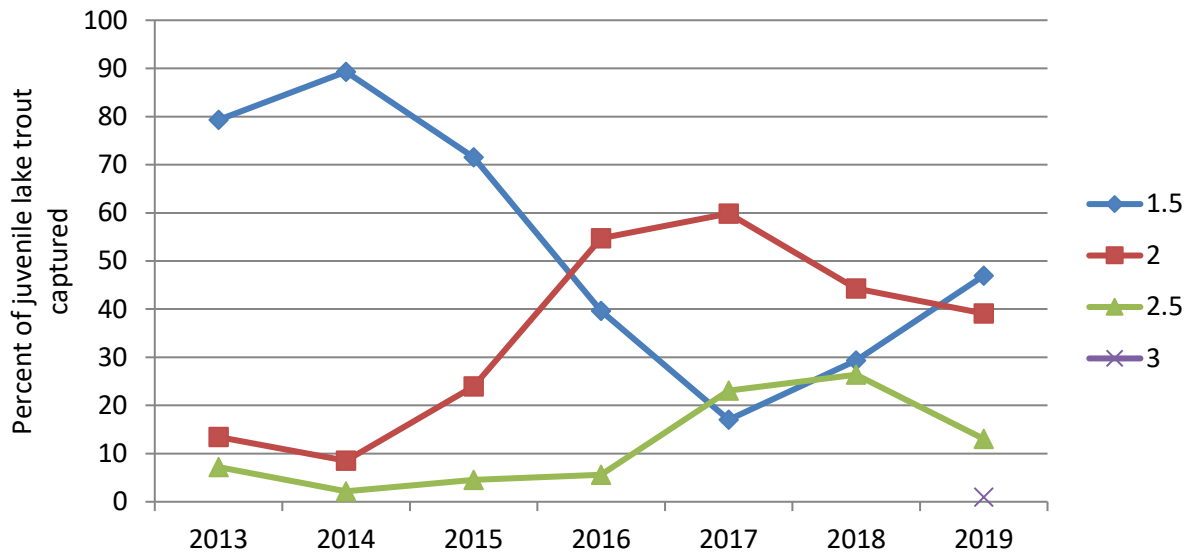


Figure 10. Percent of total juvenile lake trout catch in the spring attributable to each net mesh size in Quartz Lake. The only difference in net mesh size was the use of 28mm mesh in 2013 in addition to the other three mesh sizes. 28mm mesh was dropped beginning in 2014.

Fall

Fall netting in 2019 removed a total of 71 adult-sized (>450mm) lake trout of which 63 were mature (Figure 11). We also captured a total of 16 bull trout during the fall removal efforts (Figure 11). Fall lake trout catch increased dramatically from 2018 to 2019, going from 39 fish to 71 fish captured with similar effort. Catch-per-unit-effort (CPUE) has been variable in recent years with 2019 being one of the highest catch rates we have seen (Figure 12). This was a disappointment but not a surprise as we were catching more sub-adult fish (350-450mm) during our angling efforts in 2017 and 2018 and we knew they would mature and begin show up on the spawning grounds around 2019. What this means for the overall suppression efforts remains uncertain, but it is likely that the lack of juvenile fish being produced in the system will catch up to the adult population and adult abundance will decline precipitously over the next 3-5 years. Based on our radio-tagging, we removed only about 45% of the spawning adults in 2019. This compares poorly to previous year's where we have removed greater than 90% of the adult spawning population in a single year. The low removal rate in 2019 may result in another strong year-class of juvenile lake trout that will need to be removed at a high rate in the next 3-5 years as well. Bull trout CPUE remains fairly stable over time and is another indicator the bull trout population remains fairly stable (Figure 12).

Mean and median size of adult lake trout has declined over time, although it is apparent that we have consistently reduced the maximum size of the lake trout captured each year during the spawner netting effort (Figure 13). The median length of adult lake trout is now approximately 550mm, down from about 650mm when we started the project. It would be expected that the average size of spawners should decline in response to heavy, size selective harvest on the adults (Coleman et al. 2000). Indeed, Fredenberg (2014) demonstrated such a decline in earlier years of the project using a cumulative length-frequency analysis. Comparison of the distribution of capture lengths in the fall over time with the peak shifting to smaller, younger fish, is another indicator of a strong year-class of first year spawners that we will need to contend with over the next several years (Figure 14). It is also possible we are starting to see a compensatory response in the lake trout population from our heavy removal efforts resulting in fish maturing at earlier ages and smaller sizes. Evaluation of adult sex-ratio over time reveals little about the progress of the program (Figure 15). Male lake trout were more abundant in the catch in most years with the exception of 2014 and 2015 when we captured and removed more females. The sex ratio in 2019 was heavily skewed to males. Based on the work of Fredenberg (2014), male lake trout in Quartz Lake don't begin to mature until age-7, with females maturing a year or

two later. This could result in the skewed sex ratios on the spawning grounds we are observing as a higher proportion of males than females reach maturity in any given year. Based on the maturity schedule and the pattern of declining juvenile catch over time, we anticipate seeing a decline in adult recruitment beginning sometime from 2020-2022.

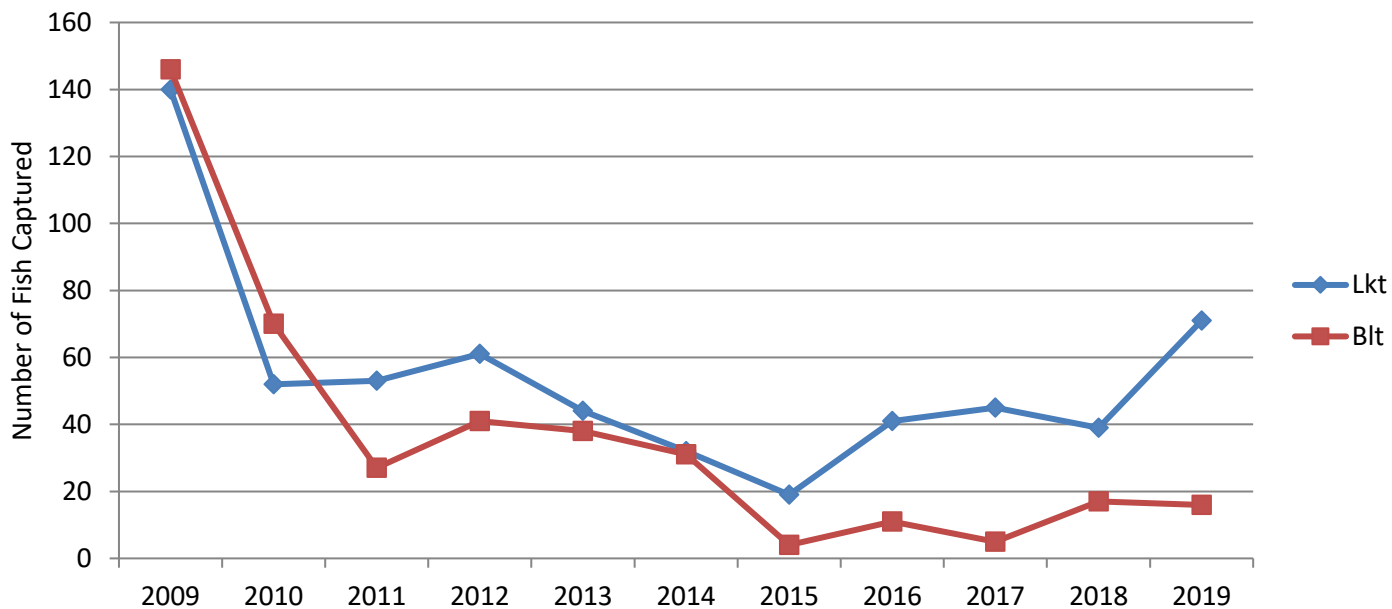


Figure 11. Total number (all adult size mature, immature, and unknown status) of lake and bull trout caught during fall adult removal gillnetting (2009-2019) in Quartz Lake, Montana.

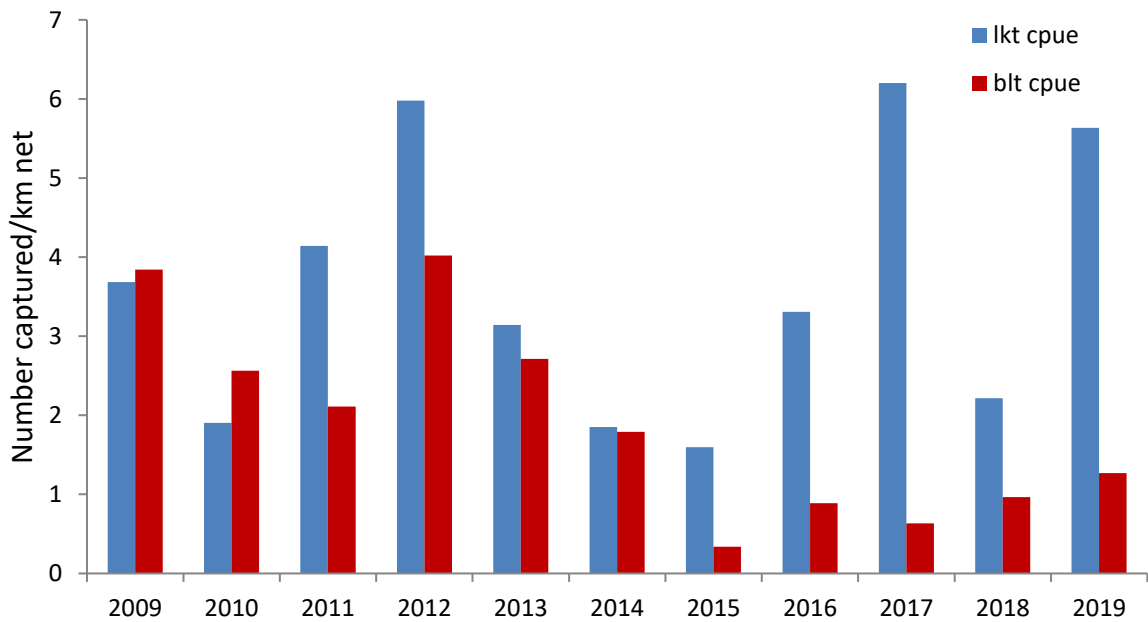


Figure 12. Catch-per-unit-effort for lake and bull trout captured during fall removal efforts in Quartz Lake, Montana.

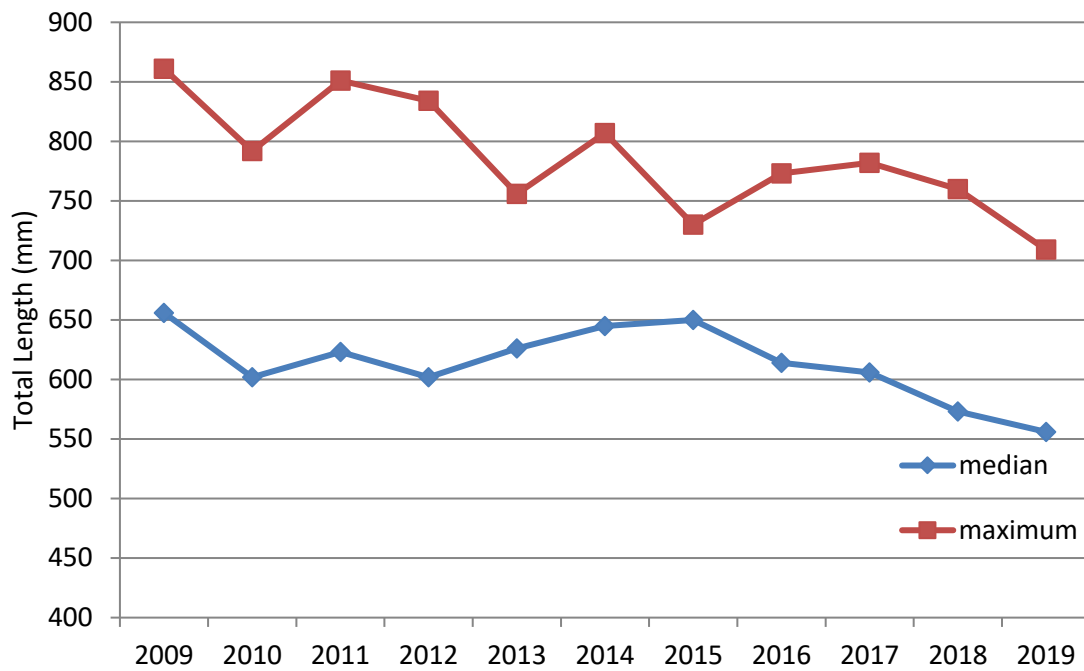


Figure 13. Maximum and median lengths of adult lake trout captured during fall removal efforts over time in Quartz Lake, Montana.

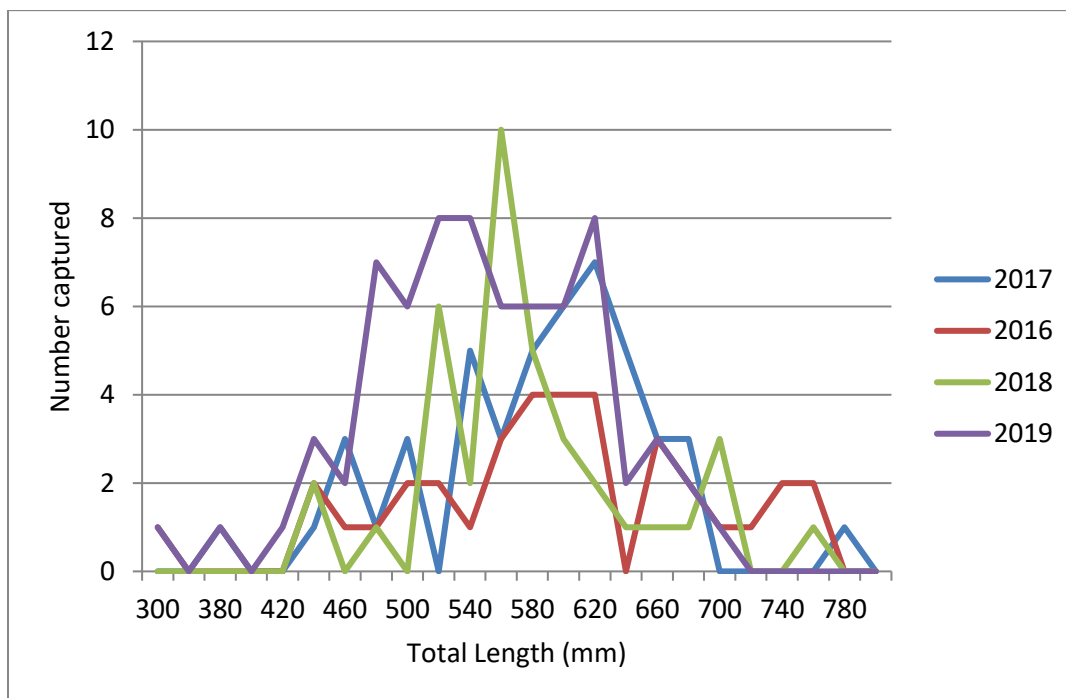


Figure 14. Length distribution of lake trout removed during fall netting over time.

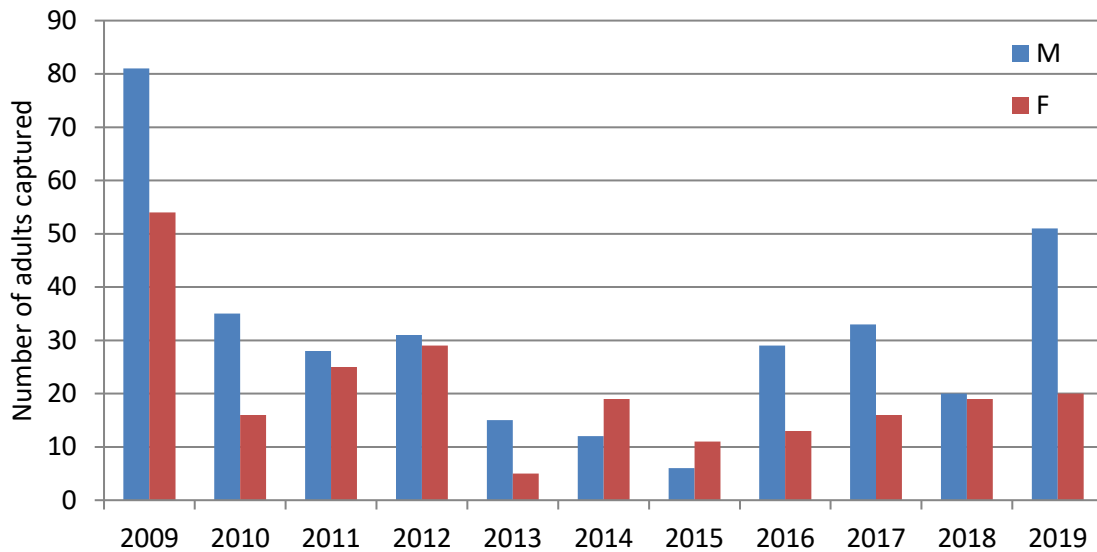


Figure 15. Sex ratio of adult lake trout captured in spawning areas over time in Quartz Lake, Montana.

Bycatch

Bycatch of non-target native species remains a concern for the project and is closely monitored. Bull trout redd counts have continued on an annual basis to monitor for any negative project impacts to the bull trout population. Total number of redds counted in upper Quartz Creek in 2019 was 27. Evaluation of the redd count data over time suggests the population remains relatively stable (Figure 16), despite some handling (and direct mortality) during netting and egg take operations in recent years.

An adaptive management approach has been used to decrease the number of bull trout and other species incidentally caught and to decrease overall by-catch mortality. To minimize bull trout bycatch mortality, soak times for spring and fall netting have been kept relatively short (2-4 hours) to give inadvertently captured native fish the best chances of survival. Although gill-net mesh size and effort has generally been standardized to allow year to year comparisons, we have made slight modifications over the years to try to reduce bull trout bycatch. An example of this includes the removal of the 45 mm and 51 mm bar measure gillnets in 2011. Analysis from 2009 and 2010 revealed that these mesh sizes accounted for the greatest number of adult bull trout mortalities, while 57 mm, 64 mm, and 70 mm bar mesh gillnets accounted for the vast majority of the adult lake trout catch.

In 2014 all of our juvenile gillnet meshes were changed from the standard 0.20 to 0.15 twine diameter. The intent was to increase the catch rate of juvenile lake trout. However, this finer mesh has also increased the catch rate of juvenile mountain whitefish. Netting data over time revealed that our bycatch of native fish is inversely correlated with water depth. Juvenile nets are fished in the deeper waters of Quartz Lake (>40m) to avoid most native species, but large schools of juvenile whitefish are still periodically encountered at these depths.

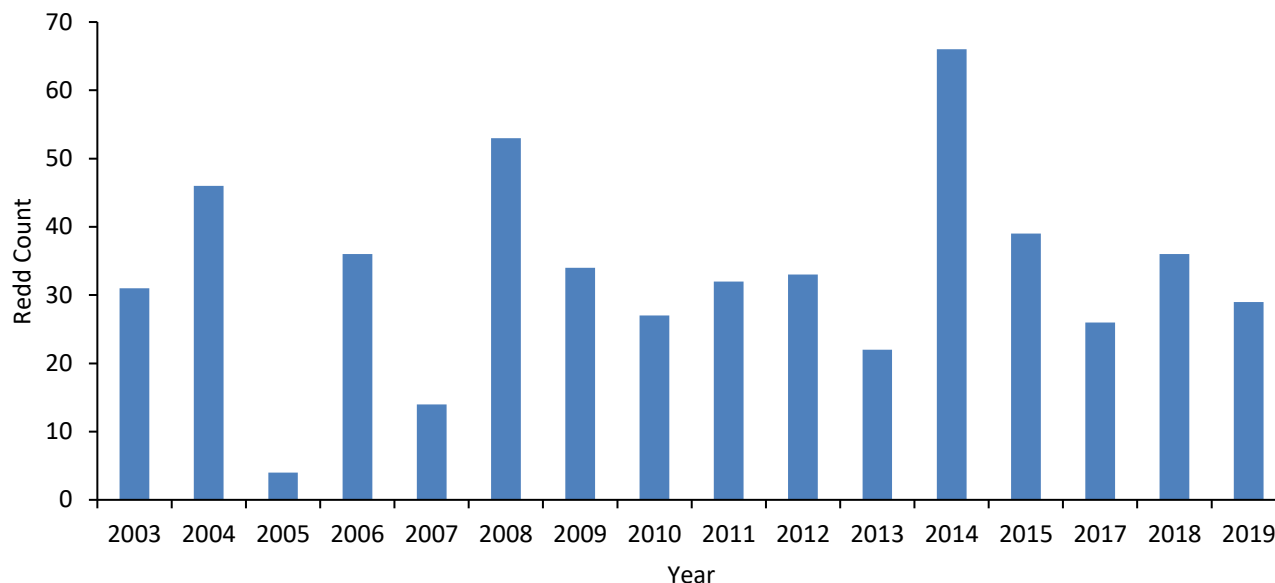


Figure 16. Bull trout redd counts from 2003 through 2019 in Quartz Creek upstream of Quartz Lake, Glacier National Park, Montana. *Redds were obscured by high flows in 2005 (Meeuwig and Guy 2007) and a count was not possible due to high water in 2016.

A total of 35 bull trout were captured via gillnetting in 2019, 19 (54%) of which were captured during spring and 16 (46%) during fall. Of the 35 bull trout captured, 13 were classified as adults ($\geq 450\text{mm}$), 8 were classified as juveniles or subadults, and the remaining 14 were not classified. Additionally, 25 adult-sized bull trout were captured during spring angling. All were released alive, but one was in poor condition and likely died after release. A total of two adult bull trout were incidentally killed in gillnetting efforts in 2019 (5.7% of total bull trout captured). No juvenile bull trout were killed in netting efforts in 2019. By caught bull trout mortality continues to be low as we become more effective at targeting lake trout while limiting capture of native species (Table 3). We will continue to look for ways to reduce mortality and bycatch of bull trout and other native species in 2020.

Bycatch of non-target species in 2019 was similar to the previous five years (Table 4). Bycatch mortality varied by species and season of capture. We captured 99% of the total mountain whitefish catch in our spring nets. Due to the longer set times, the pressure differential encountered while being pulled up from deep waters ($>40\text{m}$), and the difficulty in removing the small and fragile young mountain whitefish from the nets, the vast majority of the whitefish captured died as a result of the netting. Both sucker species were also captured but generally tolerate the netting fairly well as we observed few direct mortalities in both spring and fall nets. Most westslope cutthroat trout captured in the spring were released alive and were in decent condition considering being caught in gill nets. Some have obvious bruising and some descaling, but most likely survive following release. It has been extremely valuable to have a live well on the boat as native species can be placed in it while pumping fresh, highly oxygenated water through it to revive them.

Table 3. Annual bycatch and bycatch mortality for juvenile, adult, and unknown maturity status (no length measurement taken and immediately released) bull trout in Quartz Lake, Montana for years 2009-2019.

Year	Bycatch	Mortality
2009	38, 108, 0	14, 13, 0
2010	28, 95, 0	13, 30, 0
2011	10, 46, 0	3, 8, 0
2012	20, 45, 0	3, 9, 0
2013	50, 38, 0	22, 7, 0
2014	24, 34, 0	6, 5, 0
2015	9, 11, 0	1, 1, 0
2016	18, 20, 0	5, 4, 0
2017	5, 9, 0	1, 2, 0
2018	15, 14, 10	9, 2, 0
2019	7, 19, 9	0, 2, 0

Table 4. Annual bycatch of bull trout (BLT), westslope cutthroat trout (WCT), mountain whitefish (MWF), largescale sucker (LSS), and longnose sucker (LNS) during spring (fall) netting events from 2009-2019 in Quartz Lake, Montana.

Year	BLT	WCT	MWF	LSS	LNS
2009	--(146)	--(28)	--(498)	--(2)	--(351)
2010	53(70)	11(33)	185(55)	24(52)	216(249)
2011	29(27)	0(7)	100(17)	26(24)	285(27)
2012	24(41)	0(35)	124(51)	18(30)	143(74)
2013	52(38)	8(29)	229(25)	88(48)	237(18)
2014	27(31)	3(37)	1,237(21)	112(40)	135(41)
2015	16(4)	4(22)	690(1)	49(4)	153(5)
2016	27(11)	14(38)	1,208(11)	68(10)	563(21)
2017	11(5)	16(26)	875(8)	123(17)	218(14)
2018	22(17)	7(65)	864(3)	68(43)	347(8)
2019	19(16)	11(55)	476(7)	55(36)	319(42)

Conclusion

There are some encouraging signs in the data collected to date which suggest our netting efforts are negatively impacting adult lake trout abundance and juvenile recruitment. Catch of juvenile lake trout continues to decline with little sign of new recruits. Juvenile lake trout are vulnerable to our netting gear from about age-3 to age-5 and the decline in catch indicates a much smaller population of juveniles. The reduction in juvenile lake trout abundance is a result of the removal of adults (impacting lake trout spawning success) as well as the annual removal of juveniles. Mature lake trout abundance declined from 2009-2015 but has been slowly increasing since then as evidenced by an increasing trend in our annual fall gill net catch, as well as increased catch rates in our spring angling and increasing mark-recapture population estimates (Figures 6, 7, 11, and 12). The increase in adult abundance is likely the result of insufficient juvenile suppression and adult removal efforts occurring early in the project implementation timeline (2009-2012). We will need to intensify our juvenile and adult removal efforts going forward to overcome this increase.

Further, estimates of the number of breeding adults in the population (N_b), based on genetic material collected from juveniles removed in the program, is consistent with our observed numbers of spawners removed each year. That is, genetic data suggest an estimated N_b of <60 breeding adults for the years 2007-2010 (G. Luikart, University of Montana, personal communication).

Of concern is our continued ability to capture large lake trout during the fall spawner netting effort, some of whom based on their size (600-700mm) have likely avoided our nets in previous years and have spawned successfully. We will continue to use telemetry to search for new lake trout spawning areas in Quartz Lake so they can be effectively targeted for suppression before becoming fully established.

Fredenberg (2014) and Fredenberg et al. (2017) concluded the project needed to increase exploitation rates to achieve further reduction in lake trout abundance, which prompted the NPS to purchase and fly in a new boat in 2016 specifically designed for gill netting lake trout. This has clearly improved our ability to remove juvenile lake trout in the spring by setting much more net. The impact of the new boat is not as large in the fall sampling when we are setting much smaller amounts of net on highly localized areas. However, the new boat provides a much safer platform to work from during both spring and fall. We have documented avoidance behavior in lake trout during the fall spawner netting which impacts our ability to remove adults. Lake trout catch rates decline over time during fall efforts when we net the same areas night after night to the point where we have to take a couple of nights off to rest the spawning areas so the lake trout will return and be vulnerable to our netting once again.

Westslope cutthroat trout angler catch rates remain high (2 fish/hr. in 2019) with fish generally in excellent condition (Figure 17). Adult bull trout abundance, based on redd count data, appears to be stable. Overall, it appears as though we are successfully meeting our objective of maintaining a stable bull trout population (and westslope cutthroat trout population) through removal of lake trout. Although most monitoring metrics are trending in the right direction, others are not. Based on results to date and trends in the data, it will likely take an additional 5-10 years of netting to fully understand the impacts the program has had on both native and non-native fish populations in Quartz Lake.

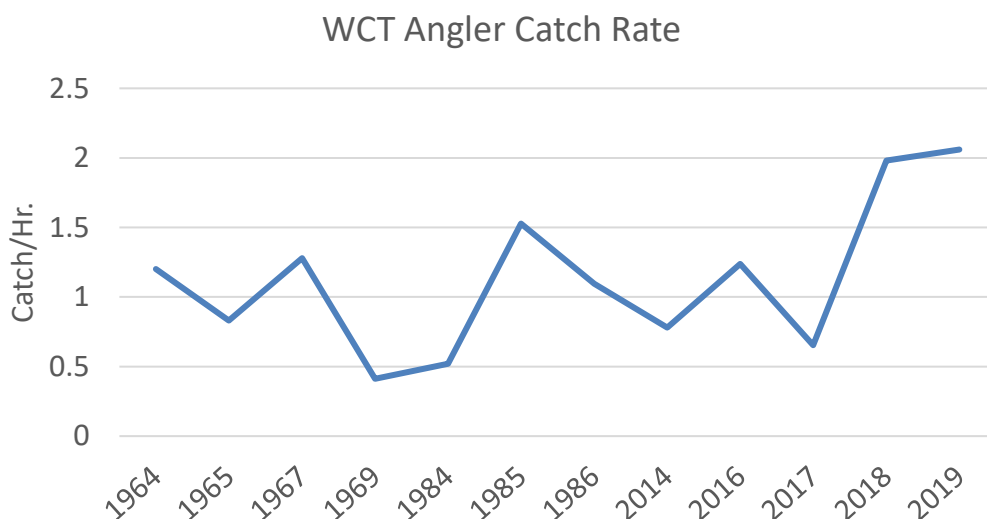


Figure 17. Westslope cutthroat trout angler catch rates in Quartz Lake over time.

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Appendix A. 2019 Telemetry Locations of Adult and Juvenile Lake Trout in Quartz Lake

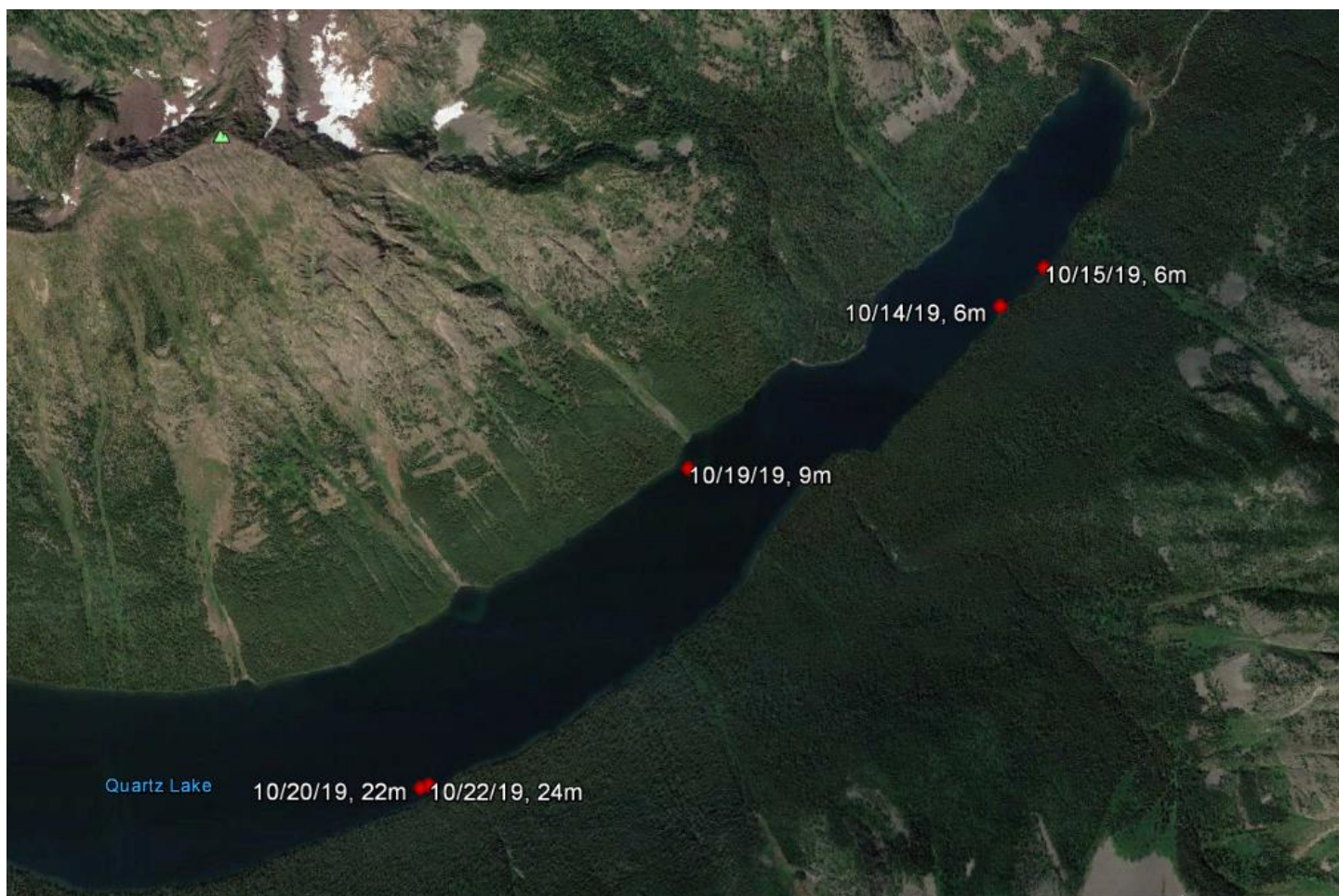


Figure A.1. 2019 Telemetry relocations for adult lake trout number 18 in Quartz Lake. Captured in spawner net and removed on 10/24/2019.



Figure A. 2. 2019 Telemetry relocations for adult lake trout number 19 in Quartz Lake.



Figure A.3. 2019 Telemetry relocations for adult lake trout number 20 in Quartz Lake.



Figure A.4. 2019 Telemetry relocations for adult lake trout number 24 in Quartz Lake. Captured and removed in spawner net on 10/12/2019.



Figure A.5. 2019 Telemetry relocations for adult lake trout number 108 in Quartz Lake. Captured and removed in spawner nets on 10/12/2019.



Figure A.6. 2019 Telemetry relocations for adult lake trout number 113 in Quartz Lake.



Figure A.7. 2019 Telemetry relocations for adult lake trout number 123 in Quartz Lake.

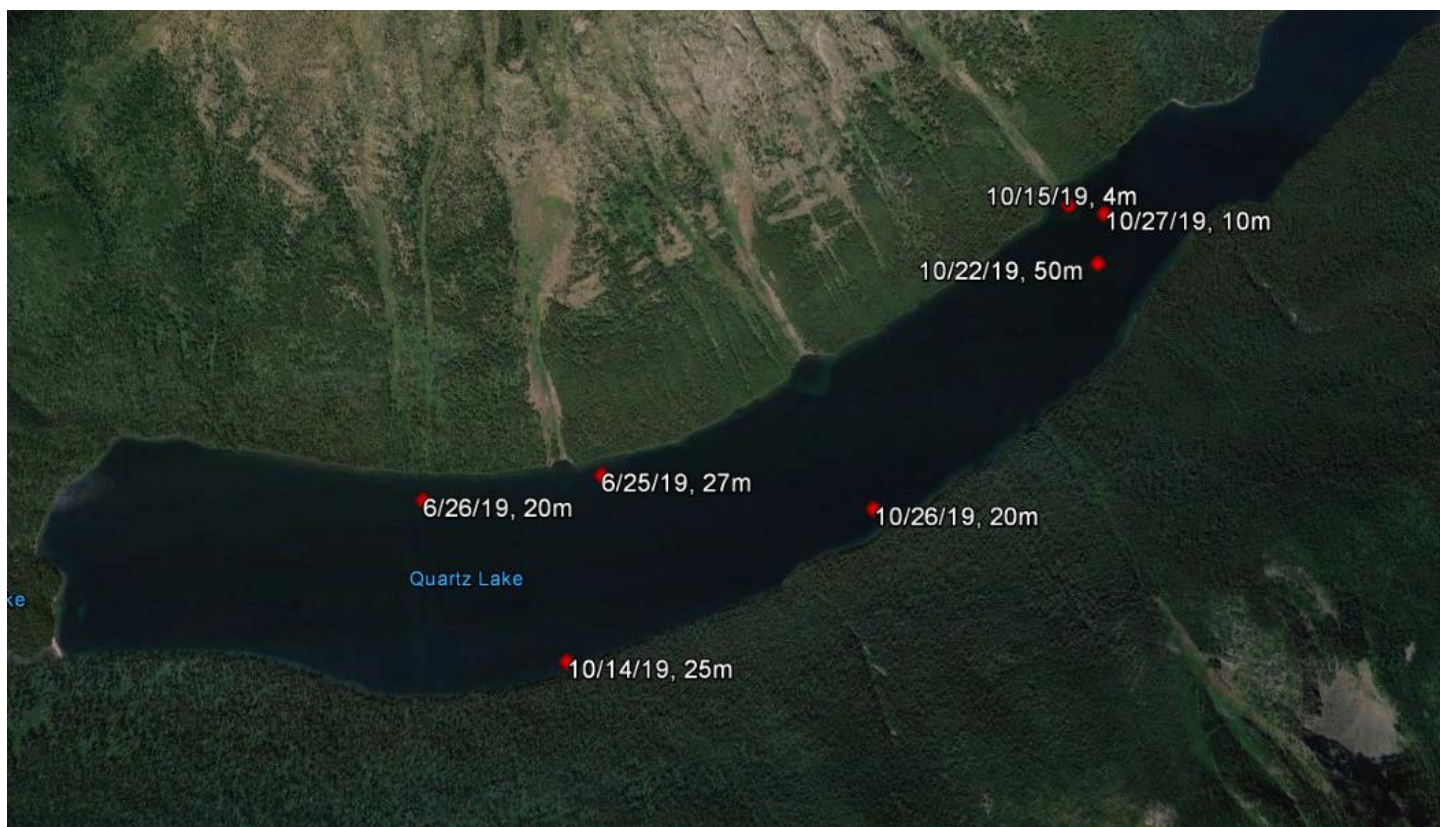


Figure A.8. 2019 Telemetry relocations for adult lake trout number 124 in Quartz Lake.



Figure A.9. 2019 Telemetry relocations for adult lake trout number 125 in Quartz Lake.



Figure A.10. 2019 Telemetry relocations for adult lake trout number 126 in Quartz Lake. Captured and removed in spawner nets on 10/10/2019.



Figure A.11. 2019 Telemetry relocations for juvenile lake trout number 04 in Quartz Lake.



Figure A.12. 2019 Telemetry relocations for juvenile lake trout number 05 in Quartz Lake.



Figure A.13. 2019 Telemetry relocations for juvenile lake trout number 06 in Quartz Lake.



Figure A.14. 2019 Telemetry relocations for juvenile lake trout number 07 in Quartz Lake.



Figure A.15. 2019 Telemetry relocations for juvenile lake trout number 08 in Quartz Lake.

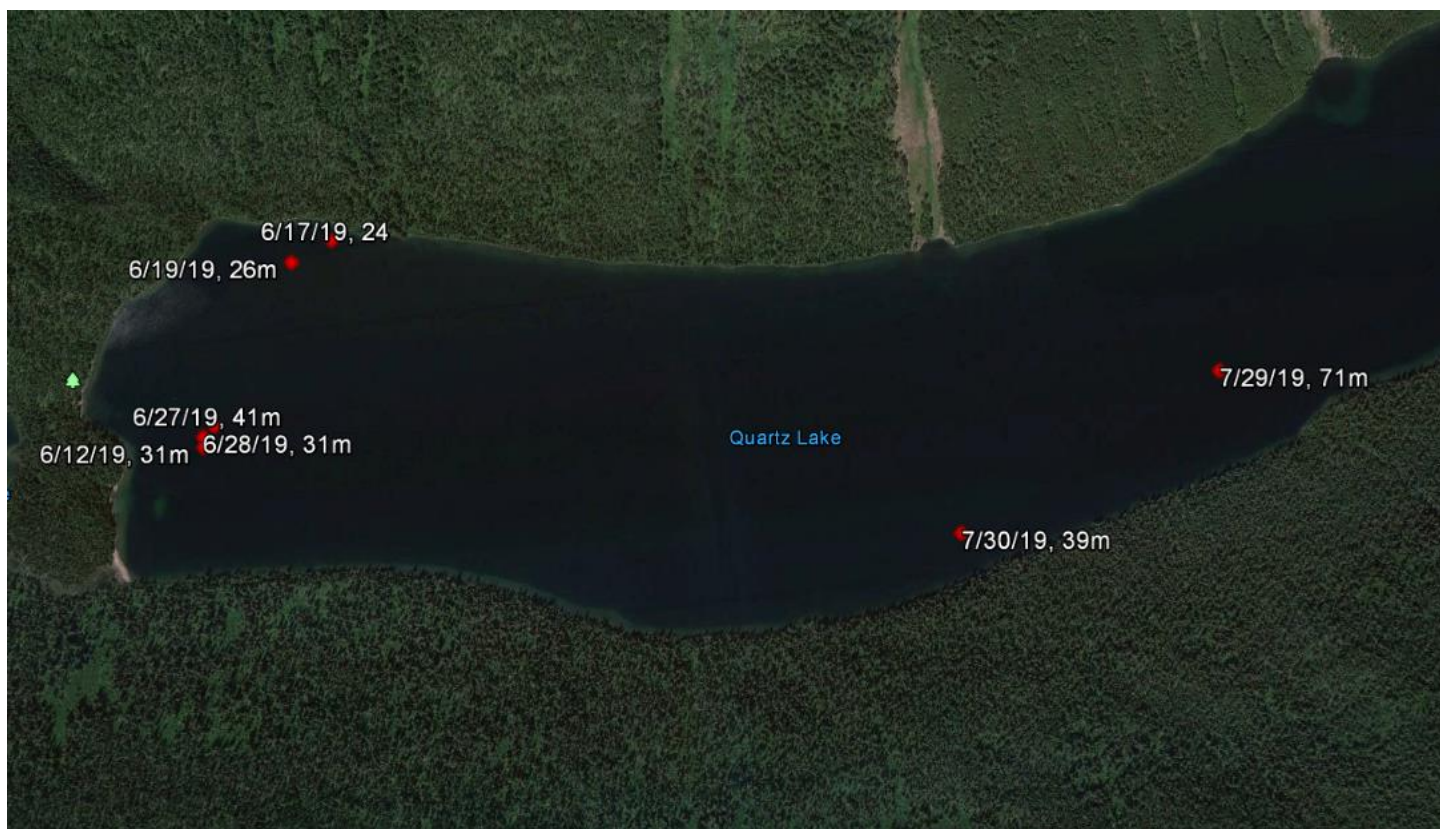


Figure A.16. 2019 Telemetry relocations for juvenile lake trout number 09 in Quartz Lake.

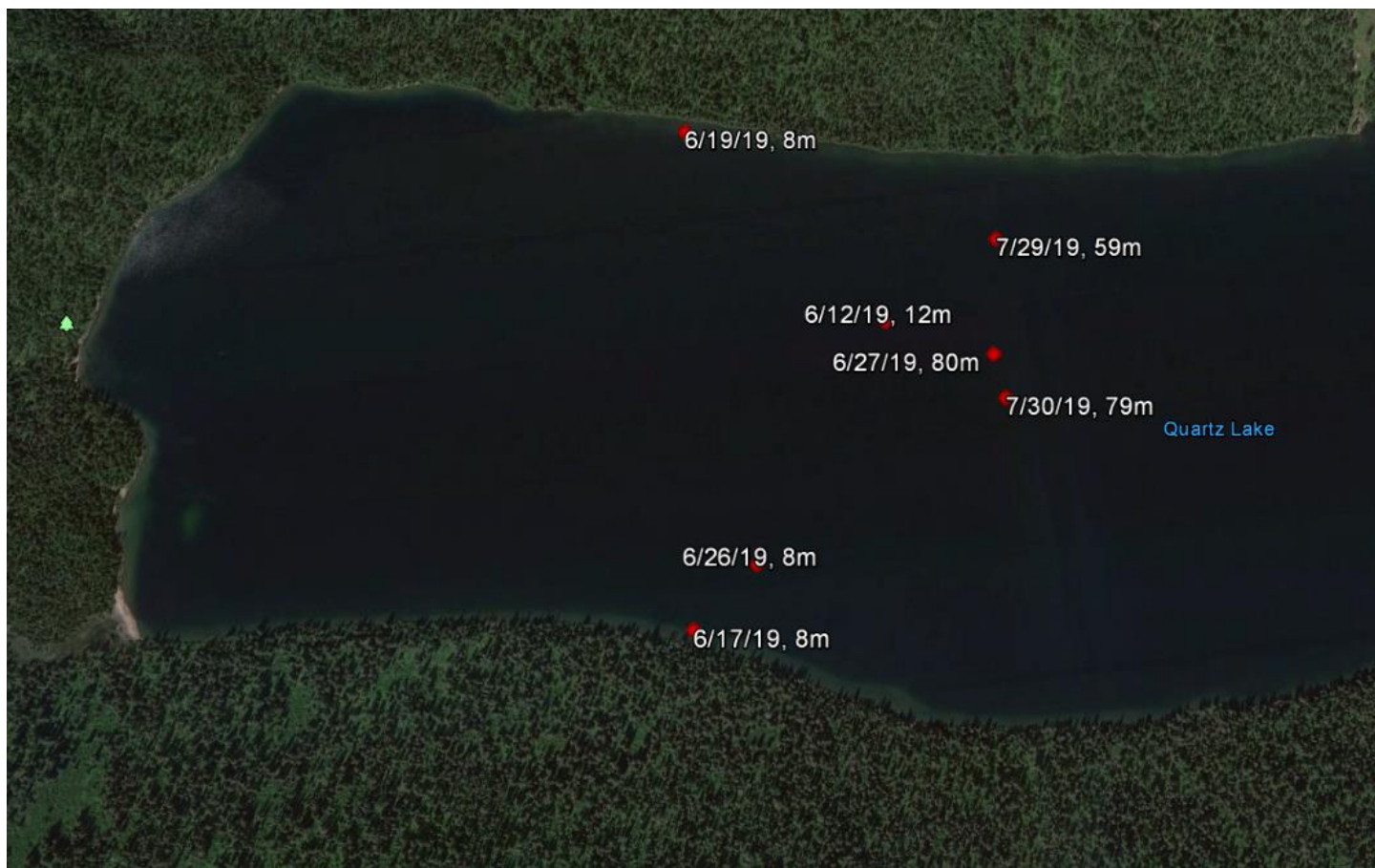


Figure A.17. 2019 Telemetry relocations for juvenile lake trout number 10 in Quartz Lake.

Native Fish and Lake Fisheries Population Monitoring

ABSTRACT

In 2009, Glacier National Park (GNP) began development of a monitoring program for native salmonids inhabiting park streams. The intent of the program is to establish baseline abundance levels in a variety of park waters to serve as useful benchmarks for monitoring changes in populations over time. We continued this work through 2019, sampling in three waterbodies. We conducted snorkel surveys on Mineral (Middle Fork Flathead drainage) and Upper McDonald (North Fork Flathead drainage) creeks. We noted presence or absence along with relative abundance of amphibians in our surveys. Numbers of fish counted during snorkeling efforts was low compared to previous years. More years of snorkel surveys are needed to assess a trend in these populations. The most significant threat to native fish on the west side of the park comes from invasive species, such as rainbow, brook, and lake trout. However, management and conservation of native fish in the North and Middle forks of the Flathead River remains complicated due to the presence of migratory and resident populations of native salmonids and expanding distribution of invasive fish species. We sampled Kintla Lake, Bowman Lake and Lake McDonald in GNP in 2019 to characterize and assess fish communities and trends occurring within the fish communities over time. We used sinking multi-filament experimental mesh gill nets, set overnight, to sample the lakes from July 10 to July 12. These lakes were considered thermally stratified during this time. We sampled Gunsight Lake in GNP in 2018 and Lost Lake in GNP in 2019 to assess the entirely invasive fish communities therein and whether there is potential to eliminate these populations in the future.

INTRODUCTION

Glacier National Park (GNP), located in northwest Montana, represents some of the most pristine and biologically diverse habitat for plants and animals found in the Intermountain West. Sitting at the core of the Crown of the Continent Ecosystem, GNP provides a diversity of stream and lake habitats for aquatic species. GNP covers approximately 1,000,000 acres, providing high-quality lentic and lotic fish habitat. GNP supports over 700 perennial lakes/ponds, ranging in size from less than an acre, up to Lake McDonald, covering almost 7,000 surface acres. GNP also provides over 2,200 km of high-quality perennial stream habitat for aquatic species. A diversity of native and introduced fish species inhabits park waters (Table 1; Table 2).

GNP encompasses the headwaters of three continental scale watersheds (Figure 1). The western portions of the park drain into the Pacific Ocean via the Columbia River, the southeastern portions of the park drain into the Atlantic Ocean via the Mississippi River, while the northeastern portions of the park drain into both the Arctic and the North Atlantic oceans via the Hudson Bay Drainage.

In order to effectively manage fishery resources and understand how landscape level changes impact these resources, data on species abundance and distribution is needed. Limited quantitative, repeatable historic fisheries data exists, and most of it was collected decades ago by Montana Fish, Wildlife, and Parks in the Flathead River drainages of the park (Read et al. 1982, Weaver et al. 1983). There were also a couple of efforts in the late 1960's and 1970's (NPS files, unpublished data) to conduct standardized gill-net surveys in some of the large lowland lakes in the North Fork Flathead Drainage of the park by the USFWS, and these data have served as useful baseline to evaluate fish species composition changes over time (e.g. lake trout versus bull trout). Later work by Mogen and Kaeding (2004) on bull trout populations in the St. Mary drainage represented the start of limited, but more intensive monitoring efforts for bull and westslope cutthroat trout in that drainage.

Establishing baseline data sets will be key to understanding how populations are responding to a changing climate and the associated changes in water temperatures, storm frequency, runoff timing, increase fire frequency, etc. Developing population estimate sections spread across the park should be valuable to monitor changes over time in fish community composition and abundance in response to expanding non-native fish species populations and climate change. This report represents a summary of data collected during 2016 field season.

Although GNP represents some of the last best wild areas in North America, recent studies have demonstrated that GNP is not immune to anthropogenic impacts such as exotic species and airborne contaminants. Recent fisheries studies (Marnell et al. 1987, Fredenberg 2002, Hitt et al. 2003, Muhlfeld et al. 2009) have demonstrated the ecological damage and genetic consequences associated with non-native fish species in park waters. For example, Fredenberg (2002) documented the replacement of bull trout as the top predator by non-native lake trout in sympatric lakes on the west side of the park during the last several decades, and lake trout are continuing to expand their numbers and range on the west side of the park (Downs et al. 2015). In response to the lake trout threat the park has recently initiated collaborative efforts to secure remaining at-risk bull trout populations. Monitoring data from park lakes will be key in evaluating any future shifts in fish community structure that may result from additional impacts of non-native species, climate change, or other habitat perturbations.

The purpose of the experimental gill net sampling was to repeat long-term trend monitoring in Kintla, Bowman and McDonald Lakes of GNP. We also sampled Gunsight Lake and Lost Lake to gain information on invasive species in these lakes. This information will be used to assess the status of native and non-native fish populations in park lakes over time and to aid managers in preserving the native species existing within the pristine waters of GNP. Temperature loggers were deployed in 2018 and 2019 to record stream temperatures in streams where they have been deployed annually since 2009.

Table 1. Native (N) and introduced (I) salmonids in Glacier National Park.

Species	Columbia Drainage	Missouri Drainage	Hudson Bay Drainage
Arctic grayling <i>Thymallus arcticus</i>	--	--	I
Brook trout (ebt) <i>Salvelinus fontinalis</i>	I	I	I
Bull trout (blt) <i>S. confluentus</i>	N	--	N
Kokanee <i>Oncorhynchus nerka</i>	I	--	--
Lake trout <i>S. namaycush</i>	I	I	N
Lake whitefish <i>Coregonus clupeaformis</i>	I	--	N
Mountain whitefish (mwf) <i>Prosopium williamsoni</i>	N	N	N
Pygmy whitefish <i>P. coulteri</i>	N	--	N
Rainbow trout (rbt) <i>O. mykiss</i>	I	I	I
Westslope cutthroat trout (wct) <i>O. clarkii lewisi</i>	N	N	N
Yellowstone cutthroat trout <i>O. c. bouvieri</i>	I	I	I
Fathead minnow <i>Pimephales promelas</i>	--	--	--
Northern pikeminnow <i>Ptychocheilus oregonensis</i>	N	--	--
Peamouth <i>Mylocheilus caurinus</i>	N	--	--
Redside shiner <i>Richardsonius balteatus</i>	N	--	--

Table 2. Native (N) and introduced (I) non-salmonids in Glacier National Park.

Species	Columbia Drainage	Missouri Drainage	Hudson Bay Drainage
Longnose sucker (lms) <i>Catostomus</i>	N	N	N
Largescale sucker (lss) <i>C. macrocheilus</i>	N	--	--
White sucker (wsu) <i>C. commersoni</i>	--	N	N
Deepwater sculpin <i>Myoxocephalus thomsoni</i>	--	--	N
Mottled sculpin <i>Cottus bairdi</i>	--	N	N
Slimy sculpin <i>C. cognatus</i>	N	--	--
Shorthead sculpin <i>C. confusus</i>	N	--	--
Spoonhead sculpin <i>C. ricei</i>	--	--	N
Burbot <i>Lota</i>	--	--	N
Northern pike <i>Esox Lucius</i>	--	--	N
Trout-perch <i>Percopsis omiscomaycus</i>	--	--	N
Lake Chub <i>Couesius plumbeus</i>	--	N	N

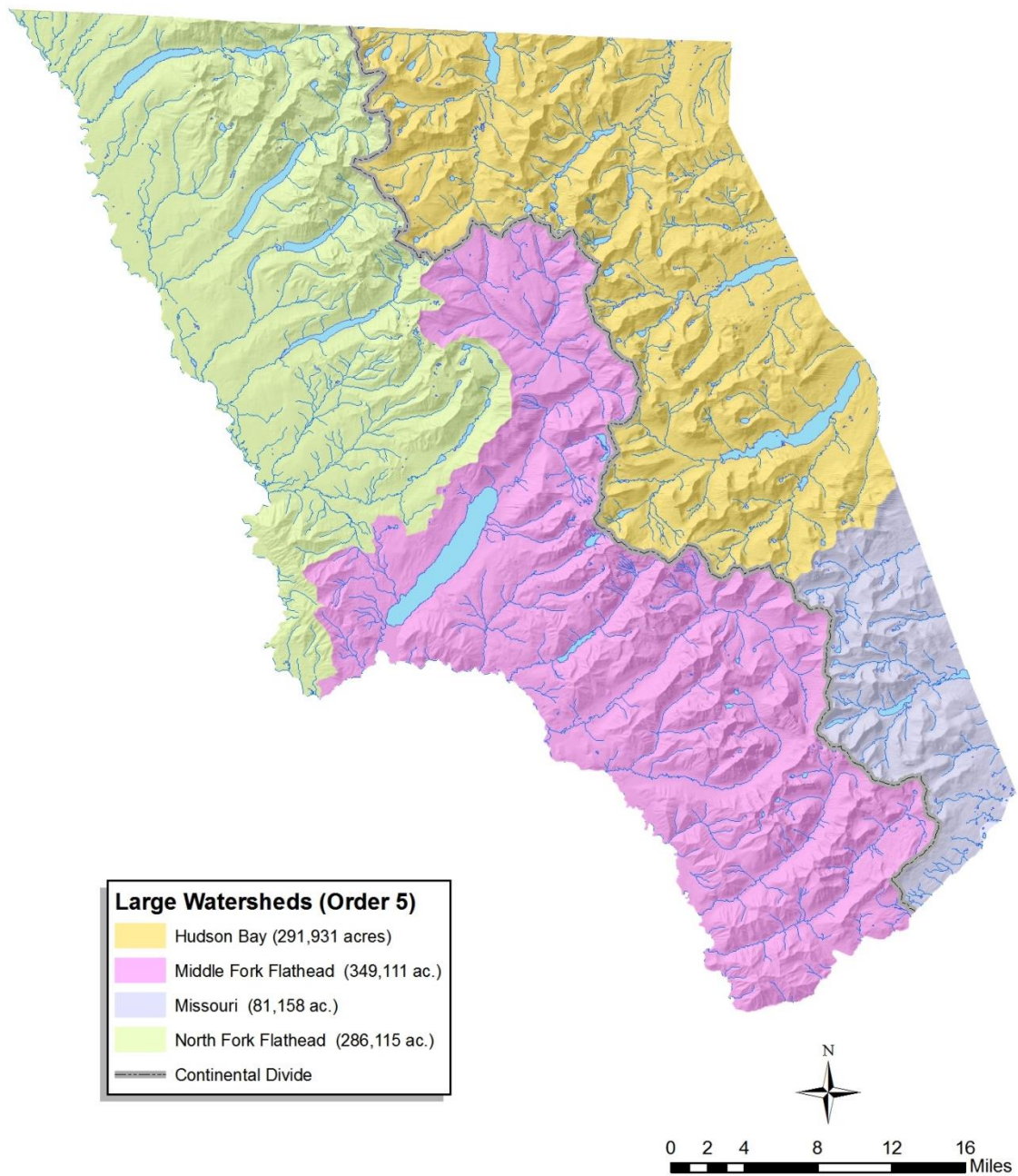


Figure 1. Major watersheds of Glacier National Park, Montana.

METHODS

Snorkeling

We used snorkeling to assess species composition and size structure. In 2018 we sampled Mineral Creek, Upper McDonald Creek, Ole Creek and Akokola Creek. In 2019 we sampled Mineral Creek, Upper McDonald Creek, Muir Creek, Ole Creek, Akokola Creek and Starvation Creek (Figure 2, Table 3). Surveying for fish using standardized snorkeling techniques has long been understood to be a valid method to survey abundance, species composition, size structure, and habitat use (Thurow 1994). We defined three size classes of fish; Class I: <49mm TL, Class II: 50-149mm TL, Class III: >150mm. Surveyors snorkeled in an upstream direction. At least one observer walked alongside the snorkeler measuring stream width, recording time snorkeler was surveying, species found, and to watch for safety hazards. Fish were not counted in the survey until the snorkeler had moved past them in order to avoid duplicate counting of fish. Snorkel estimates involved identifying a representative reach of stream approximately 100-300 m long. We used a high-gradient riffle break or other natural drop in the stream channel as section starting and ending points. When possible, snorkel surveys were performed within previously sampled electrofishing or historical (e.g. Read et al. 1982) snorkeling sections.

Due to the cold water temperatures generally found in the park, the timing of our sampling, and previous length-frequency data, we assumed bull trout (blt) ≥ 60 mm and westslope cutthroat trout (wct) ≥ 45 mm were age-1 and older for estimation of abundance and catch-per-unit effort (CPUE). This was confirmed by examining the length-at-capture data from our 2009 field sampling. These assumptions are further supported by other studies. In some cold systems similar to GNP, wct fry may not even emerge from the gravel until mid-August (Scarnecchia and Bergersen 1986, Downs 1995). Scarnecchia and Bergersen (1986) indicated few cutthroat trout from headwater systems in Colorado exceeded lengths of 30-35 mm before they entered their first winter. Therefore, using a lower limit (i.e. 45 mm) as a cutoff for inclusion in estimates of age-1 and older wct is more appropriate for most park waters containing rearing wct. Fishes of these sizes (blt ≥ 60 mm and wct ≥ 45 mm) can be efficiently viewed with snorkeling gear and thereby provide for estimation of abundance.

For snorkeling surveys, GPS coordinates of the upstream and downstream end of the sections were recorded in UTM using the NAD 83 datum. Digital photographs were taken of the upstream and downstream ends of each section, and wetted widths were measured at approximately 10m intervals to calculate the wetted stream area sampled. Stream temperature was recorded in Celsius at the time of sampling with a handheld thermometer. Dominant and sub-dominant substrate sizes for the entire reach were estimated visually, using six general size classes of bed material: bedrock, boulder, cobble, gravel, sand, and silt. The presence and relative abundance of amphibians was noted as present-common or present-rare.

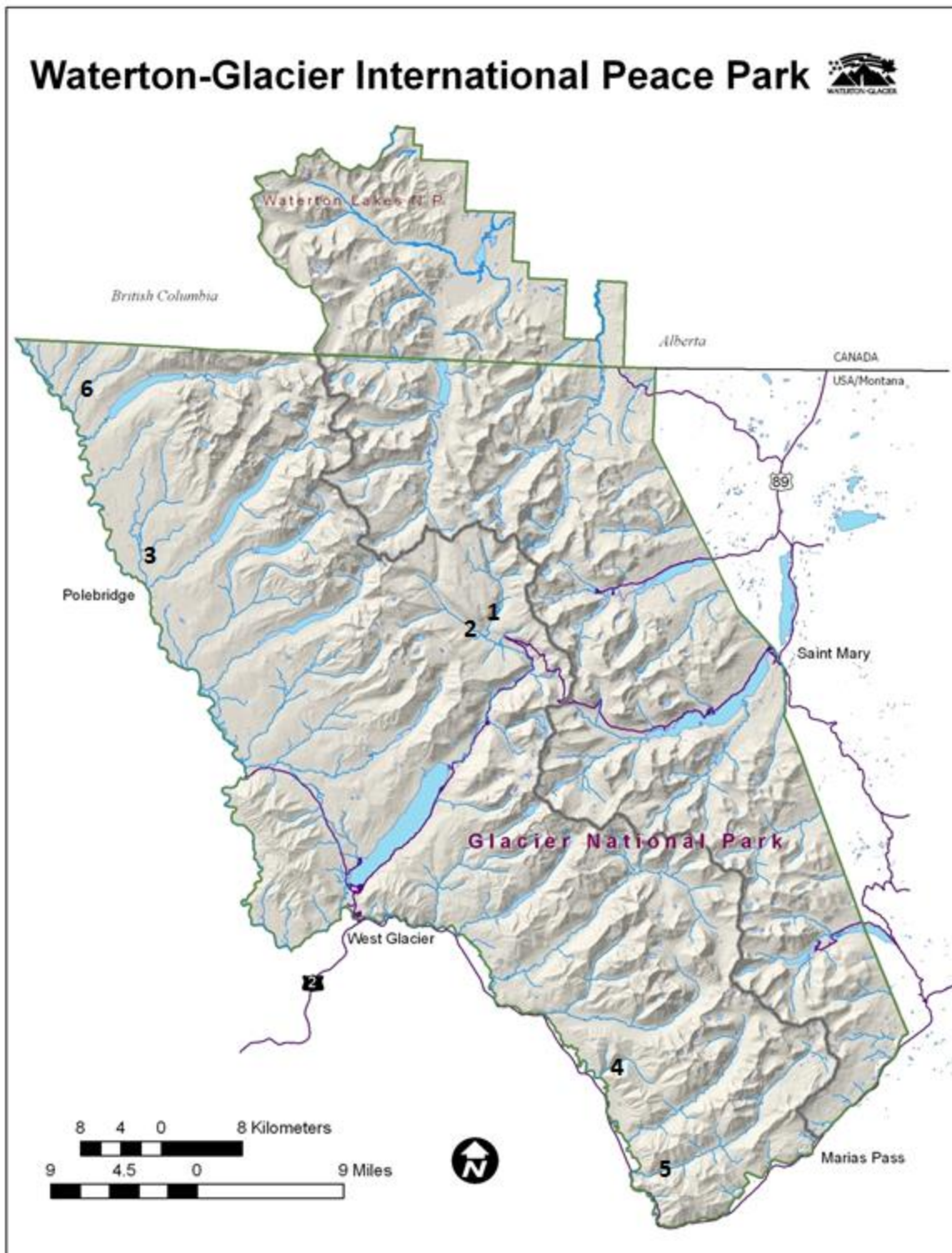


Figure 2. Stream sampling sites for snorkeling and gillnetting efforts in GNP, Montana 2019. Site numbers from Table 3.

Table 3. Location information for native fish population monitoring efforts in 2018 and 2019.

Major Drainage	Water name (Site No.)	Method	Location of downstream end of reach (NAD 83, UTM)	Dates sampled	Sample Reach Length (m)
Middle Fork Flathead	Mineral Creek (1)	Snorkel	12U 0303790, 5360495	7/26/2018, 8/21/2019	377
	Upper McDonald Creek (2)	Snorkel	12U 292335, 5403907	7/23/2018, 8/21/2019	329
	Muir Creek (4)	Snorkel	12U 303844.5, 5360679.8	8/22/2018, 8/20/2019	150
	Ole Creek (5)	Snorkel	12U 307427.1, 5350703.3	8/23/2018, 8/19/2019	200
North Fork Flathead	Akokola Creek (3)	Snorkel	11N E0698772, N5409951	8/15/2018, 8/22/2019	270
	Starvation Creek (6)	Snorkel	11U 691070.2, 5423938.4	8/21/2018	200

Gill Netting

We used sinking multi-filament experimental mesh gill nets to assess species composition and size structure in park lakes (Figure 3). Nets were generally distributed around the entire length in the same locations as previous netting (Table 4). Nets were 38.1 m (125') X 1.83 m (6') experimental multifilament nylon with five 7.62 m (25') panels consisting of 19 mm (3/4"), 25 mm (1"), 32 mm (1-1/4"), 38 mm (1-1/2") , 51 mm (2") bar measured mesh. Thirty-pound lead core and polyfoam line was used to hold the net upright, while maintaining contact with the lake bottom. Gunsight Lake was sampled in 2019. Bowman Lake, Kintla Lake, Lincoln Lake and Lost Lake were sampled in 2019. We attempted to repeat set locations and depths used historically to monitor the lake fish communities except for in Lost Lake as it was the first year that we conducted experimental gill netting there (Fredenberg et al. 07; Table 1). Where historic sampling locations were unavailable, we came up with new set locations by dispersing net sets around the circumference of the lake to produce a more lake wide consistent sample.

Gillnets were set in late afternoon/early evening around the circumference of each lake and were allowed to soak overnight. They were retrieved at sunrise the following day. Nets were set with the smallest mesh towards shore. All bull trout captured alive that did not physically display brook trout characteristics were counted in the net catch total, estimated in length and were immediately released. Bull trout that displayed any distinctive hybrid characteristics or had expired prior to net removal were sexed, maturity status was determined, genetic samples were collected, and otoliths were removed and collected. All other fish were measured (TL; mm), weighed (g) and carcasses returned to the lake. In 2019, USFWS personnel worked with us to collect fish tissues to test for disease in all species sampled in Kintla and Bowman Lakes.

Catch-per-unit effort (CPUE), defined as the number of fish captured per net-night, was used as the primary measure of catch rate over time. We calculated average length and weight for each species to facilitate comparisons of size structure between and within species populations both currently and over time within each lake.

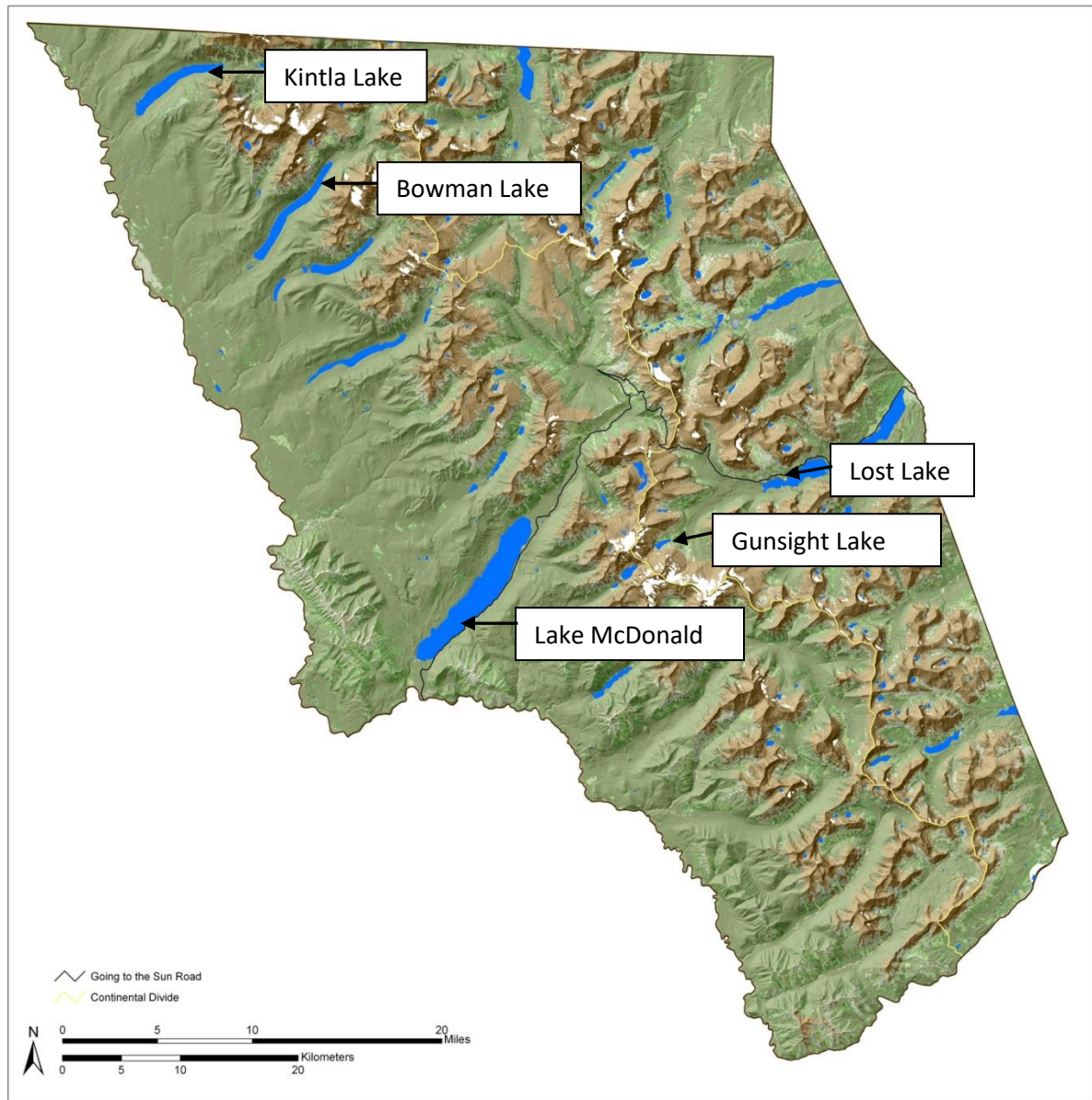


Figure 3. Location of lakes netted in 2018 and 2019 in Glacier National Park.

Table 4. Gillnet locations and set and retrieval dates for Gunsight Lake, Bowman Lake, Kintla Lake and Lake McDonald in GNP in 2018 and 2019.

	Net	UTM Zone	UTM Easting	UTM Northing	Set Date	Retrieve Date
Gunsight Lake	1	12	300160.2	5389375	8/7/18	8/8/2018
	2	12	299583.4	5388983.8	8/7/18	8/8/2018
	3	12	299345.8	5388756.4	8/7/18	8/8/2018
	4	12	299954.1	5389243.3	8/7/18	8/8/2018
	5	12	299143.7	5389089.6	8/7/18	8/8/2018
	6	12	299350.4	5389153.5	8/7/18	8/8/2018
	7	12	299859.7	5389452.5	8/9/18	8/10/2018
	8	12	300102.2	5389528.4	8/9/18	8/10/2018
Bowman Lake	1	11	0707196	5415910	7/29/2019	7/30/2019
	2	11	0707371	5414681	7/29/2019	7/30/2019
	3	11	0706600	5414339	7/29/2019	7/30/2019
	4	11	0708277	5416474	7/30/2019	7/31/2019
	5	11	0709371	5418108	7/30/2019	7/31/2019
	6	11	0710974	5420710	7/30/2019	7/31/2019
	7	11	0709944	5417904	7/30/2019	7/31/2019
	8	11	0705538	5412826	7/29/2019	7/30/2019
	9	11	0707149	5414111	7/29/2019	7/30/2019
	10	11	0707950	54116140	7/30/2019	7/31/2019
Kintla Lake	1	11	6945640	54246480	7/23/2019	7/24/2019
	2	11	6948150	54249520	7/23/2019	7/24/2019
	3	11	6954380	54259050	7/23/2019	7/24/2019
	4	11	6981280	54279210	7/22/2019	7/23/2019
	5	11	0701250	54285820	7/22/2019	7/23/2019
	6	11	0700098	54277730	7/22/2019	7/23/2019
	7	11	0697265	54263910	7/22/2019	7/23/2019
	8	11	0696726	54257060	7/22/2019	7/23/2019
	9	11	0695785	54248520	7/22/2019	7/23/2019
	10	11	0697107	54272390	7/22/2019	7/23/2019
Lake McDonald	1	12	0280372	53796950	7/15/2019	7/16/2019
	2	12	0279965	53813070	7/15/2019	7/16/2019
	3	12	0283160	53848710	7/15/2019	7/16/2019
	4	12	0285743	53885320	7/16/2019	7/17/2019
	5	12	0286332	53899860	7/16/2019	7/17/2019
	6	12	0288558	53905310	7/17/2019	7/18/2019
	7	12	0287582	53884440	7/16/2019	7/17/2019
	8	12	0286501	53859560	7/16/2019	7/17/2019
	9	12	0282950	53822750	7/15/2019	7/16/2019
	10	12	0280617	53819490	7/15/2019	7/16/2019

Lost Lake Trap and Gill Netting

Lost Lake is a roughly 5-acre lake located upstream on the north side of Saint Mary Lake and about 100m on the north side of the going to the sun road. In the spring, it is directly connected to the Saint Mary Lake by a small, roughly ½ mile long high-gradient stream. Non-native brook trout (*Salvelinus fontinalis*) are the only species known to exist in this lake and were planted there around the middle of the 20th century. The source of inlet flow comes from snowmelt and groundwater. The outlet channel is characterized by high-gradient cascades forming a natural upstream fish migration barrier. To assess brook trout abundance prior to any management actions, we conducted a mark-recapture estimate using hook and line and gill nets. Anglers caught and marked brook trout with a fin clip and staff also deployed mini trap nets to capture brook trout for marking in early July. The recapture effort consisted of overnight gill net sets using sinking 1.5-inch fine mesh, green tint monofilament gill nets.

RESULTS AND DISCUSSION

Snorkeling

Mineral Creek

Mineral Creek is a second order tributary to McDonald Creek in the McDonald Lake drainage. The downstream end of the sampling site is located at the trail crossing (bridge) which is located ~350 m upstream of the confluence of Mineral Creek and upper McDonald Creek.

Based on the size structure of the population and the location of the sampling site in the drainage it is likely this population is resident. Mineral Creek contains genetically pure wct based on samples collected in 2008.

In 2018 and 2019 wct was the only species of fish observed in this reach of Mineral Creek. All size groups of wct were observed (Table 5). Thirty-one wct were observed by snorkelers in 2018 (7/26/2018) resulting in a density estimate of 0.54 wct/100 m². Mineral Creek was again snorkeled on 8/21/2019 and 10 wct were observed, resulting in a wct population density of 0.26 wct/100m². It is difficult to say whether a decrease in densities observed are due to changes personnel conducting snorkel surveys, natural changes in the stream reach or due to natural fluctuations in the population. Additional years of sampling will help us better understand any trends in abundance.

Large cobble was estimated to be the dominant substrate type followed by gravel in the reach in both 2018 and 2019. Average wetted-width measured 12.3m in 2018 and 10.24m in 2019. Water temperature was measured at 13.5°C in 2018 and 12.0°C in 2019. Temperature was recorded at the time of sampling in both years. No amphibians were observed in Mineral Creek in 2018 or 2019.

A comparison of snorkel data collected in this reach of Mineral Creek from 2015-2019 shows a declining trend of wct in all three length groups (Figure 4). However, additional years of snorkel data should give a better representation of abundance and size structure and counts could have been impacted by fluctuating water temperatures and flows amongst years as well as different personnel conducting the counts between years.

Table 5. Size class, species, count and fish per 100m² for fish observed during snorkel surveys in Mineral Creek in 2018 and 2019.

Species	Lengths	2018	2019
WCT	≤45 mm	11	2
	46-149 mm	18	6
	≥150 mm	2	2
Age 1+ /100m2		0.54	0.26

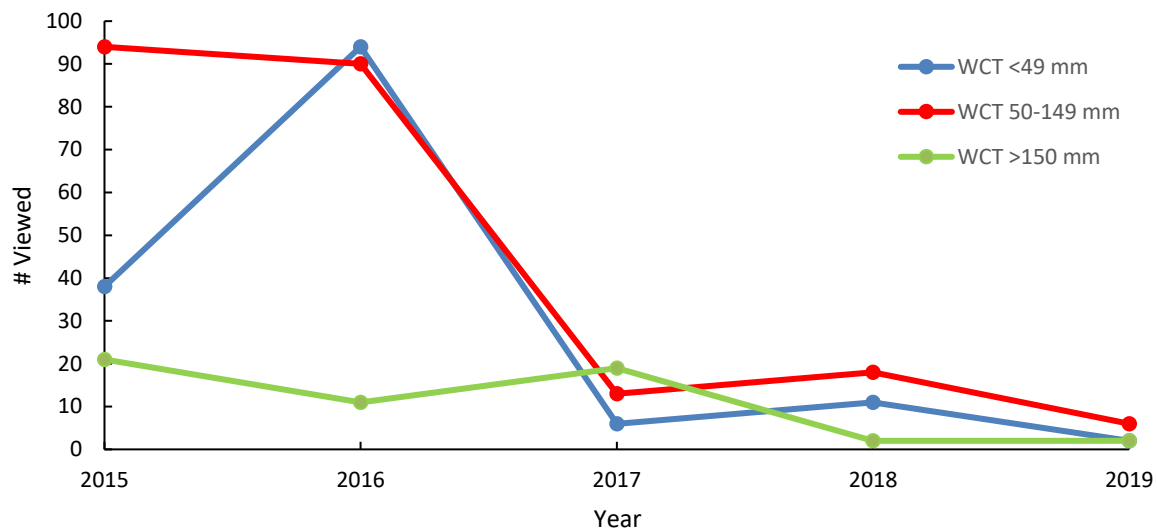


Figure 4. A comparison of numbers of wct in three different length groups observed during snorkel surveys conducted in Mineral Creek and their associated trends. Sampling was conducted in 2015 - 2019.

Upper McDonald Creek

Upper McDonald Creek above the confluence with Mineral Creek is a third order tributary to the M. Fk. Flathead River (Figure 2). It flows primarily southeast, until it joins Mineral Creek at which time it flows southwest to Lake McDonald. Upper McDonald Creek drains approximately 28,700 ha. In 2013 and 2015-2019 we snorkeled from the confluence with Mineral Creek upstream for 329m, the elevation at the sampling site was approximately 1,147m. Sections below Lake McDonald have been surveyed previously by Weaver et al. (1983), however, we do not believe that others have snorkeled above the Mineral Creek confluence. McDonald Creek upstream of McDonald Falls is known to contain genetically pure wct (NPS, unpublished data).

In all sampling years from 2013 to 2019 wct were the only species of fish observed in this reach of McDonald Creek. Upper McDonald Creek contains a relatively low density of cutthroat trout which is not unexpected given the harsh natural environment and low productivity of the system. Densities have varied over time (Table 6, Figure 5), with 2013 and 2019 representing the extremes from high density to low. There is no angler harvest of westslope cutthroat trout on the west side of the park and very few, if any, anglers are likely to fish mineral creek. The variability observed likely represent the natural fluctuation in abundance in this section driven by variables such as precipitation timing. Observer variability also likely contributes to the overall variability in fish numbers observed from year to year.

Cobble was estimated to be the dominant substrate type followed by gravel in the reach. Average wetted-width measured 11.43 m in 2013, 10.78 m in 2015, 11.8 m in 2016, 13.6 m in 2017, 13.7 m in 2018, and 11.5 m in 2019. Water temperature was measured at 13 °C in 2013, 12 °C in 2015, 13 °C in 2016, 13 °C in 2017, 12 °C in 2018, and 11.4 °C in 2019. No amphibians were observed in McDonald Creek during snorkel surveys.

Table 6. Size class, species, and count for fish observed during snorkel surveys in McDonald Creek.

Species	Length Class	2013	2015	2016	2017	2018	2019
WCT	≤45 mm	11	18	6	5	16	1
	46-149 mm	31	19	16	9	16	0
	≥150 mm	23	3	1	3	0	0
	Age 1+ /100m2	1.44	0.62	0.47	0.32	0.42	0

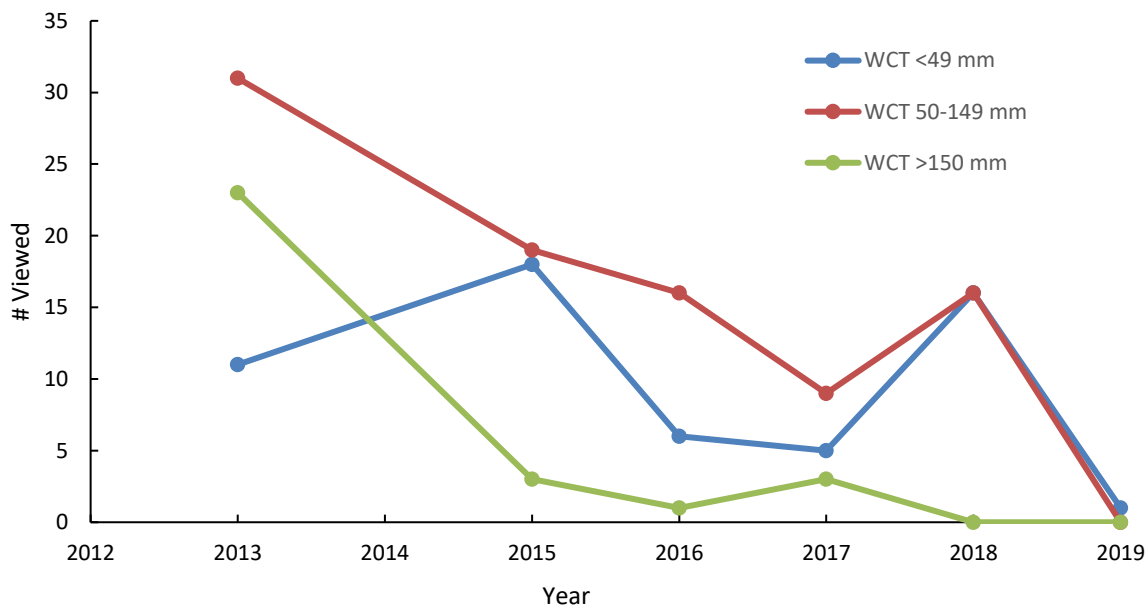


Figure 5. A comparison of numbers of wct in three different length groups observed during snorkel surveys conducted in Upper McDonald Creek and their associated trends. Sampling was conducted in 2013 and 2015-2019.

Akokola Creek

Akokola Creek is a third order tributary and flows primarily southeast and is a direct tributary to the N. Fk. Flathead River (Figure 1). We snorkeled Akokola Creek in 2015, 2018 and 2019. Akokola Creek was also snorkeled in 2004. An isolated and genetically pure bull trout population resides in Akokola Lake (which is upstream of our survey reach). Akokola Lake is isolated from upstream migrating fish due to a manmade fish barrier that was built in 2015.

Abundance of both westslope cutthroat trout and mountain whitefish observed during snorkel surveys of Akokola Creek varied from year to year (Table 7). Ried et al. (1982) reported an age-1+ wct density of 2.4/100m², which is within the range we have observed. In addition, one juvenile bull trout and one juvenile rainbow trout were observed across all sample years. Akokola Creek supports both resident and migratory westslope cutthroat trout as well as mountain whitefish and bull trout. Upper Akokola Creek tested genetically pure as recently as 2008 (NPS unpublished data), but hybrid spawner of rainbow and westslope cutthroat trout have been observed lower in the drainage (Muhlfeld et al. 2009b). In addition, sampling crews have observed fish they believed to be hybrids in electrofishing surveys conducted lower in the drainage (USGS, unpublished data)). Individual juvenile bull trout were observed in Akokola Creek in 2014 and 2018.

Cobble was estimated to be the dominant substrate type followed by course gravel in the reach in every year Akokola Creek was sampled. Average wetted-width measured in 2015 was 8.15m, in 2018 was 8.4m and in 2019 was 9.04m. Water temperature was measured at 13°C (9/2/2014) 17.5°C (8/10/2015), 9.7 °C (8/15/2018) and 13.6°C (8/22/2019). No amphibians were observed in Akokola Creek in any of the surveys.

Table 7. Size class, species, and count for fish observed during snorkel surveys in Akokala Creek. In 2018, the section length surveyed was almost double that of the other years.

Species	Length Class	2014	2015	2018	2019
WCT	≤45 mm	35	29	22	20
	46-149 mm	45	31	23	3
	≥150 mm	19	7	4	0
	Age 1+ /100m ²	5.24	2.66	1.07	0.19
MWF	≤45 mm	69	4	13	2
	46-149 mm	15	0	2	2
	≥150 mm	11	0	0	0
	Age 1+ /100m ²	2.13	0	0.13	0.13

Muir Creek

Muir Creek is a third order tributary and flows primarily south and is a direct tributary to the M. Fk. Flathead River (Figure 1). We snorkeled Muir Creek in 2013, 2014, 2018 and 2019. We are not aware of any other times that Muir Creek has been surveyed by snorkeling.

In Muir Creek in 2018 we observed 17 wct <49mm, 19 wct between 50 and 149mm and 15 wct >150mm . In 2019, we observed one wct in the <49mm length group, one wct in the 50 to 149mm length group and one wct in the >150mm length group (Table 8). In addition, there were seven unknown species of fish observed in the <49mm length group and two unknown species of fish observed in the 50-149mm length group. These are presumed to be wct as the only species present are blt or wct in Muir Creek, with wct being common and bull trout rare. Density estimates for age-1+ wct in 2018 and 2019 were 4.6/100m² and 0.92/100m² respectively. For comparison, Weaver et al. (1983) observed 11.6 age-1+ in Muir Creek approximately 30 years earlier. Survey numbers varied substantially over time with 2019 showing very low wct abundance of age-1+, and sampling in 2020 will help us further understand if this observed decline represents a trend. Some of this variability is actual variability in abundance and some is variability in counts done by different snorkelers with varied experience.

Table 8. Size class, species, and count for fish observed during snorkel surveys in Muir Creek.

Species	Length Class	2013	2014	2018	2019
WCT	≤45 mm	0	0	17	8
	46-149 mm	5	11	19	3
	≥150 mm	11	20	15	1
	Age 1+ /100m ²	1.99	5.2	4.6	0.92

Boulder/large cobble was estimated to be the dominant substrate type followed by course gravel in the reach in every year Muir Creek was sampled (2013, 2014, 2018 and 2019). Average wetted-width measured in 2018 was 4.9m

and in 2019 the average wetted width was 2.7m. This suggests either a drastic change in flow between years or that a somewhat slightly different reach may have been sampled since the personnel that conducted the snorkel survey in 2019 were new to the stream and the sample reach. In either case, wetted width measurements were much greater in 2018 than 2019. Water temperature was measured at 8.5°C in 2018 and 8.1°C in 2019. No amphibians were observed in Muir Creek in 2018 or 2019.

Ole Creek

Ole Creek is a third order tributary and flows primarily south and is a direct tributary to the M. Fk. Flathead River (Table 1). We snorkeled Ole Creek in 2018 and 2019. Results were highly variable (Table 9). Additional surveys should be completed to establish a more consistent baseline, if possible.

Cobble and coarse gravel was estimated to be the dominant substrate type followed by gravel in the reach in both years Ole Creek was sampled (2018 and 2019). Average wetted-width measured in 2018 was 8.15m and in 2019 the average wetted width was 10.3m. Water temperature was measured at 7.5°C in 2018 and 8.1°C in 2019. No amphibians were observed in Ole Creek in 2018 or 2019.

Table 9. Length class, species, and count for fish observed during snorkel surveys in Ole Creek in 2018 and 2019.

Species	Length	2018	2019
WCT	<49 mm	29	0
	50-149 mm	31	0
	>150 mm	7	2
RBT	<49 mm	0	0
	50-149 mm	1	0
	>150 mm	0	1
MWF	<49 mm	4	0
	50-149 mm	0	0
	>150 mm	0	0
BLT	<49 mm	0	0
	50-149 mm	0	0
	>150 mm	0	2

Gillnetting

Gunsight Lake

Eight (six the first evening and two the second) nets were set in Gunsight lake between 0n 8/7 and 8/9 in 2018 (Figure 6). Rainbow trout were the only species caught in Gunsight Lake (n=104) in as they are the only species present there. Catch-per-unit-effort was 13 fish per net night. The lake supports a fairly robust population of non-native rainbow trout averaging approximately 300mm (Figure 7). Consideration should be given to removing the rainbow trout and replacing them with wct, a species that is native to the St. Mary drainage. There was little fish data previously reported in Gunsight Lake so this single sampling event and the information obtained could not be compared to previous years. Genetic samples were collected for future analysis if warranted.



Figure 6. 2018 Gill net locations in Gunsight Lake, Glacier National Park.

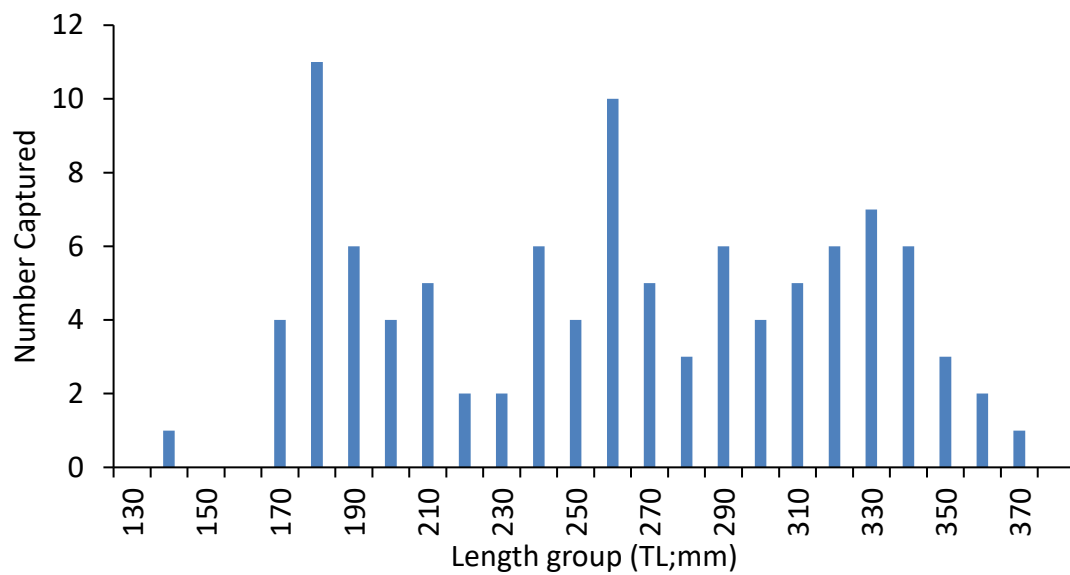


Figure 7. Length frequency of rainbow trout captured in Gunsight Lake, Glacier National Park, Montana.

Lost Lake

We used mini trap nets and hook and line methods to capture brook trout for marking during the population estimate effort. We fished seven trap net nights in early July and caught 57 brook trout. We also conducted hook and line sampling at the same time and captured another 40 brook trout. We caught a total of 609 brook trout, 447 were caught in 1.5-inch monofilament gill nets, 89 were caught in multifilament gill nets, 57 were caught in trap nets and 40 were caught via hook and line sampling. The estimated abundance of brook trout is 1,725 fish >120mm (95%CI=172-2,682). It is clear that we did not mark a large enough proportion of the population and therefore our confidence intervals are very large. However, with 20 recaptures we should have a reasonably unbiased point estimate of population abundance. Brook trout averaged 210 mm (TL) (Figure 8). The 1.5" stretch measure, green-tint, fine mesh sinking gill nets captured the widest range of brook trout sizes among all the sampling methods.

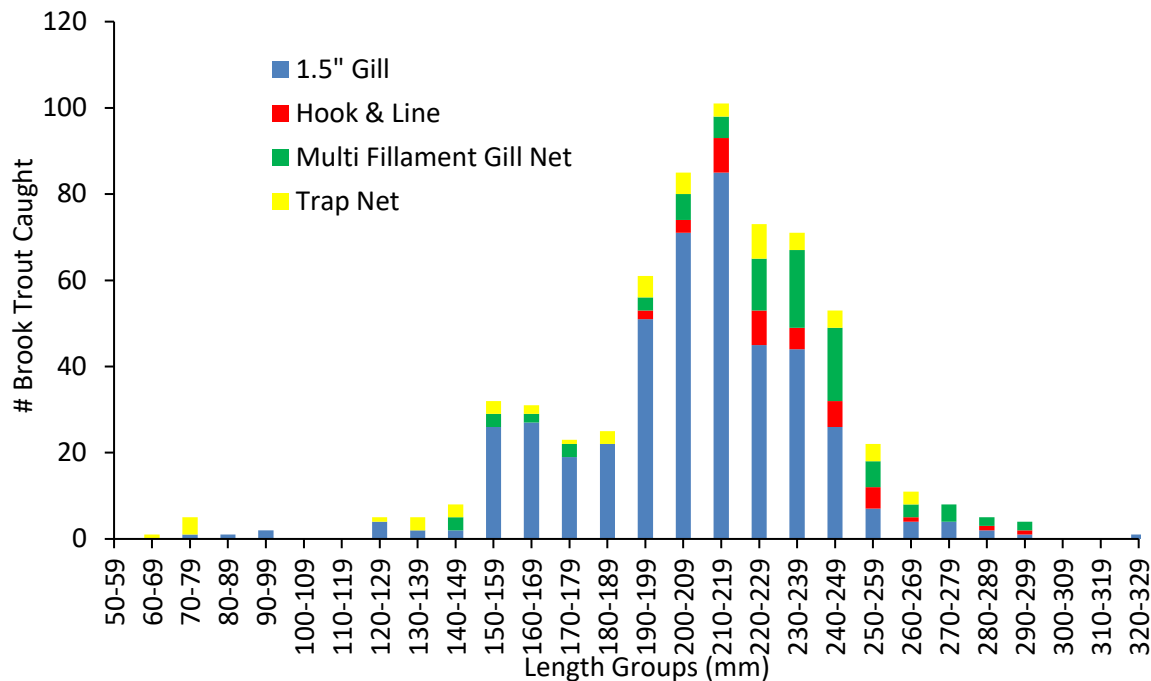


Figure 8. Length distribution of brook trout captured in Lost Lake, Glacier National Park in July and August, 2019.

Jefferson Creek Electrofishing

Jefferson Creek is a tributary to Bowman Creek and has been identified as a possible bull trout spawning stream (USGS, unpublished data). We shocked Jefferson Creek on 7/31/2019 for 856 seconds and captured two juvenile bull trout and two westslope cutthroat trout. The bull trout were both 108mm and were likely age-1. The cutthroat trout were 81 and 101mm in length and both also likely age-1 fish. The water temperature at the time of sampling was 8.6°C, and given the late July sampling date, this is likely as warm as the stream gets. Jefferson Creek is likely a spawning stream for both species. Additional surveys are warranted to identify critical areas of the stream for bull trout spawning and rearing.

Stream Temperature Monitoring

Lee Creek

We installed a thermograph in Lee Creek (Saint Mary River drainage). The thermograph on Lee Creek was located approximately 20m downstream of the Chief Mountain Highway bridge crossing. Stream temperature reached a high of 15.1 °C in Lee Creek on August 10, 2018 (Figure 9).

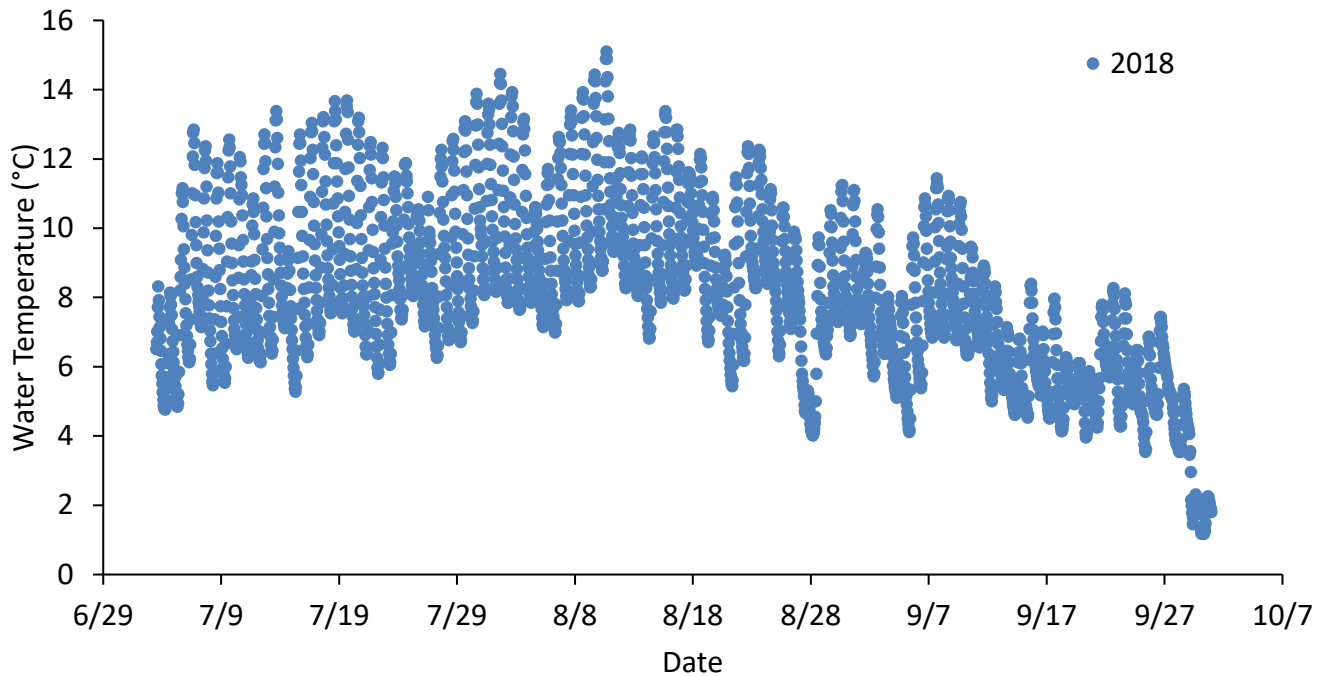


Figure 9. Lee Creek stream temperatures during 2018, Glacier National Park, Montana.

Ole Creek

We installed a thermograph in Ole Creek (Middle Fork Flathead River drainage). The thermograph on Ole Creek was located approximately 20m upstream of the suspension bridge at the trail crossing on lower Ole Creek. Stream temperature reached a high of 15.0 °C in Ole Creek on August 10, 2018 (Figure 10).

Starvation Creek

Starvation Creek is a third order tributary to the N. Fk. Flathead River (Read et al. 1982). It flows from Canada into the park prior to entering the N. Fk. Flathead River 6 km south of the Canadian border (Figure 2). Westslope cutthroat trout genetic samples were collected in 2008 and did not show any evidence of hybridization (MFWP, unpublished data). Previous researchers have documented bull trout, westslope cutthroat, mountain whitefish, and sculpin in Starvation Creek (Read et al. 1982, Downs et al. 2011).

We installed a thermograph in Starvation Creek located near the confluence with the North Fork Flathead River, about 100 m upstream of the trail crossing in 2018 and 2019. Stream temperatures peaked at approximately 18.0 °C in each year (Figure 11).

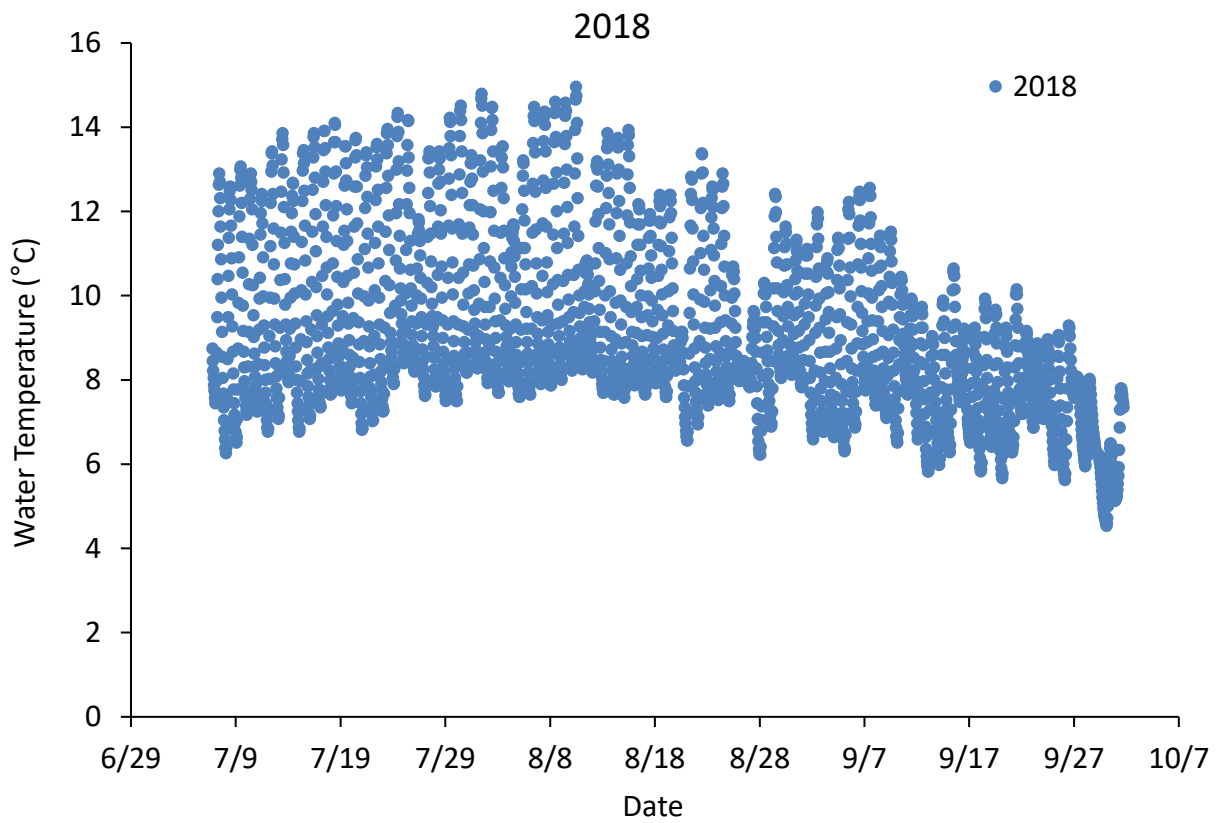


Figure 10. Ole Creek stream temperatures during 2018, Glacier National Park, Montana.

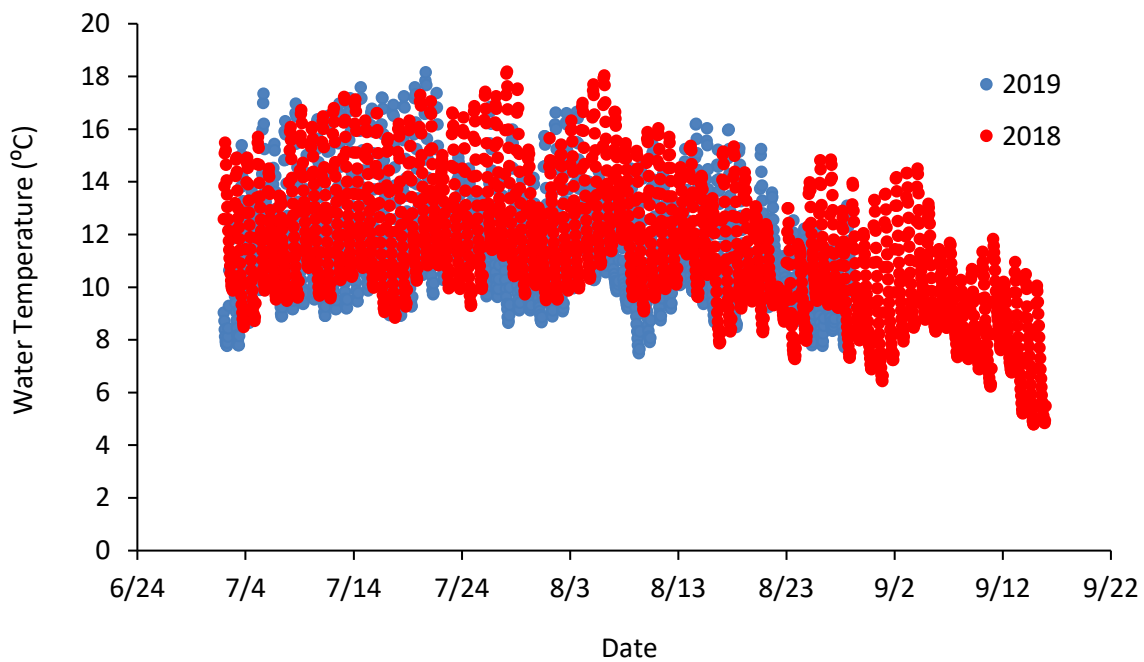


Figure 11. Stream temperature in Starvation Creek during 2018 and 2019, Glacier National Park, Montana.

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Glacier National Park Bull Trout Redd Counts

ABSTRACT

We conducted bull trout *Salvelinus confluentus* redd counts in six streams/stream reaches in Glacier National Park in 2019. We were unable to survey most of the bull trout spawning streams due to an early snowstorm and subsequent high water. In addition, in some we did survey (i.e. Park and Ole creeks) redds could not be reliably identified because the tails were flattened by high water. It was not possible to survey any streams in the St. Mary River drainage because of several feet of fresh snow fell in early October, immediately prior to the scheduled counts. We counted a total of 33 redds across the park. Quartz Lake remains one of the strongest monitored bull trout populations residing wholly within the park, with a total of 27 redds. Redd counts in 2019 for Flathead Lake migratory bull trout populations spawning in the Middle Fork Flathead River tributaries in the park were either not counted or the counts were compromised due to high flows. In addition, the NPS in partnership with both the US Fish and Wildlife Service and Montana Fish, Wildlife, and Parks have stocked 4,678 juvenile (age-0 and age-1) bull trout into Grace Lake and 2,152 age-1 bull trout into Logging Lake between 2016-2019. These bull trout came from wild eggs taken from Quartz Creek. The intent of the project is to establish a “new” bull trout population in Grace Lake that is secure from invasive lake trout and speed up bull trout recovery in Logging Lake in support of lake trout suppression efforts currently underway.

INTRODUCTION

Bull trout *Salvelinus confluentus* are one of only four native salmonids present in Glacier National Park (GNP) waters located west of the Continental Divide. They are one of six native salmonids present in GNP waters located east of the Continental Divide. GNP and the Blackfoot Nation have the unique distinction of supporting the only bull trout populations located east of the Continental Divide in the U.S. portion of their range. In addition, GNP supports both native (Hudson Bay drainage) and introduced (Columbia River drainage) populations of lake trout, occupying lake habitats along with bull trout, creating unique management challenges.

Bull trout exhibit three distinct general life-history forms – resident, fluvial, and adfluvial. Resident bull trout spend their entire lives in small tributaries, whereas fluvial and adfluvial forms hatch in small tributary streams then migrate into larger rivers (fluvial) or lakes (adfluvial). In the lakes of GNP, bull trout exhibit the adfluvial life history strategy. These bull trout grow to maturity in the lakes, and then spawn in tributaries or lake outlets. Migratory adult bull trout generally move upstream to spawning or staging areas from May through July, although some fish wait until the peak spawning time of September and October before entering spawning streams (Fraley and Shepard 1989; Schill et al. 1994; Downs and Jakubowski 2006). Spawning typically occurs in September and October in the Flathead River/Lake system (Block 1953; Fraley and Shepard 1989; Meeuwig 2008), including Glacier National Park lakes (Tennant 2010). Eggs over-winter in spawning streams until the following spring, when newly hatched fry emerge from the gravel. Age-0 bull trout can often be found in side-channels and along channel margins following emergence (Fraley and Shepard 1989). Migratory juvenile bull trout have been documented emigrating from natal streams in two pulses, with one pulse occurring in the spring with high water and the other in the fall associated with declining water temperatures and fall precipitation events (Downs et al. 2006). Juveniles may rear from one to five years in natal streams, with most emigrating at age-2 and age-3 (Downs et al. 2006). Age-0 outmigrants have been reported in some adfluvial populations, but these outmigrants did not appear to survive well to adulthood where studied (e.g. Downs et al. 2006). Resident and migratory forms may be found together, and either form can produce resident or migratory offspring.

Bull trout egg incubation success has been inversely correlated to increasing levels of fine sediment (<6.35 mm diameter) in spawning nests (redds) (Montana Bull Trout Scientific Group 1998). Spawning site selection has been related to areas of strong intragravel flow exchange (both upwelling and downwelling) (Baxter and Hauer 2000). Juvenile bull trout abundance has been positively correlated with low summer maximum water temperatures (below 14°C) and with the number of pocket pools in stream reaches (Saffel and Scarnecchia 1995). Unembedded cobble substrate is an important overwinter habitat type for juvenile bull trout (Thurow 1997; Bonneau and Scarnecchia 1998). Excess fine sediment holds the potential not only to reduce egg and embryo survival but might also limit juvenile bull trout abundance in streams by reducing the amount of interstitial spaces available for overwinter habitat. Channel stability, habitat complexity, and connectivity are all important components in bull trout population persistence (Rieman and McIntyre 1993).

Bull trout are part of a historic fish assemblage that is fundamental to the biodiversity of GNP and represent the evolutionary legacy of a top-level aquatic predator in GNP. Protecting native fish resources is a high priority for the park's conservation and management programs (NPS 2006). Ongoing research, monitoring, and management efforts conducted by GNP and its partners remain critical in understanding bull trout population dynamics in the park, and in establishing management programs to benefit native fish.

Redd counts, or spawning nest counts, are used across the range of bull trout to monitor population trends. They are typically used as an index of abundance to gauge the relative strength of adult escapement from year to year. They can also be used to estimate actual adult escapement by expanding the redd counts to fish numbers using various spawner to redd ratios. Redd counts require far less effort to conduct than other traditional monitoring methods such as trapping, and yet provide valuable information on bull trout at the watershed and/or population scale. However, redd counts are not without limitation, as the technique has been shown to be prone to observer variability and error (Dunham et al. 2001, Muhlfeld et al. 2006), yet they continue to remain an important monitoring tool for bull trout populations.

Redd counts are conducted in Glacier National Park (GNP) annually by the National Park Service (NPS), the U.S. Fish and Wildlife Service (USFWS), Montana Fish, Wildlife, and Parks (MFWP), and the U.S. Geological Survey (USGS). The longest redd count dataset on bull trout spawning activity in GNP is from three tributaries (Ole, Park, and Nyack creeks) to the Middle Fork Flathead River, associated with monitoring bull trout populations from Flathead Lake. Bull trout redd counts have been conducted annually in the St. Mary drainage on the east side of the park since 1997.

GNP is unique as it and the adjacent Blackfeet Indian Reservation are the only place where bull trout occur east of the Continental Divide in the U.S. portion of their range. GNP supports a diversity of life-history strategies for bull trout, including both resident and migratory forms. Resident bull trout have been documented in the St. Mary River drainage (Mogen and Kaeding 2004), while migratory fish from Flathead Lake use tributaries to the Middle and North forks of the Flathead River for spawning and rearing (Weaver et al. 2006). Other populations on the west side of GNP use the lake systems within the park for subadult rearing and adult residence, while spawning and rearing in upstream reaches of their inflow tributaries (e.g. Quartz Lake) (Meeuwig 2008). Less commonly, other west side populations (e.g. Upper Kintla Lake) use the lake environment for subadult rearing and adult residence, while spawning occurs in the outlet stream.

Bull trout spawning surveys were initiated by USFWS staff between 2002 and 2004 for a number of these “disjunct” west side bull trout populations (Meeuwig et al. 2007). A number of other bull trout populations on the west side of the park have not been monitored beyond recent single year electrofishing and gill net surveys (Meeuwig et al. 2007), and we simply do not know where they spawn or long-term population trends (e.g. Lincoln, Isabel, Upper Isabel lakes). It will be prudent to establish index redd count monitoring for additional populations on some frequency, as they represent the majority of “secure” populations of bull trout on the west side of GNP (Fredenberg et al. 2007).

METHODS

Experienced fisheries staff from GNP, USGS, MFWP and the USFWS identified and enumerated bull trout redds in 2019. Redd surveys generally occur during the first full three weeks of October. Surveys occurred between October 2-October 15 in 2019. Early to mid-October is the preferred time for counting bull trout redds as most bull trout spawning has already occurred (peak spawning occurs in mid to late-September), most redds are still clearly visible, and it is consistent with the timing of earlier counts.

Redds were located visually by walking along annual monitoring sections within each tributary. Redds were defined as areas of clean or “bright” gravels at least 0.3 x 0.6 m in size with gravels of at least 76.2 mm in diameter having been moved by the fish (where other fall spawning species may be present such as brook trout), and with a mound of loose gravel downstream from a depression (Pratt 1984). In areas of superimposition, each distinct depression was counted as one redd. Only disturbed areas of the streambed that observers felt were likely made by fish were classified as bull trout redds and were included in the counts (as opposed to those disturbed areas of the streambed that may have been caused by stream hydraulics). Individual redd locations were located using GPS technology where the spatial distribution of spawning activity was of particular interest.

RESULTS AND DISCUSSION

GNP, USGS, and USFWS, staff surveyed three stream reaches in the N. Fk. and three stream reach’s in the Middle Fork Flathead River drainages (Figure 1). High water compromised counts in Ole and Park creeks. The resulting counts should be viewed as minimums and not included in any trend analyses.

Because the identified spawning habitat for these populations occurs within GNP, it is largely unaffected by threats typically associated with bull trout spawning habitat in other areas of their range (i.e. road building, residential development, timber harvest). Some traditional threats do exist however, largely in the form of trespass cattle grazing in the GNP portion of the Kennedy and Lee creek drainages and the construction and operation of Sherburne Dam and

the Milk River Irrigation Project (USFWS 2002). Trespassing cattle have been observed wading in Kennedy Creek in GNP in the primary bull trout spawning area during and after bull trout spawning (J. Mogen, USFWS, personal communication), as well as in Lee Creek (C. Downs, NPS, personal communication). Recent studies (Gregory and Gamett 2009) have identified the potential for significant damage to bull trout spawning nests as a result of cattle trampling. Recent efforts by the Blackfoot Nation to fence cattle out of the bull trout spawning area on Kennedy Creek should benefit this bull trout population.

Sherburne Dam and the St. Mary Irrigation Canal impact GNP native fish populations and represent the single threat to bull trout populations face in the U.S. portion of the Hudson Bay drainage (USFWS 2002). Construction of Sherburne Dam from 1914-1921, located just outside of the GNP boundary, created Sherburne Reservoir which flooded over 8 km of shallow lake and stream habitat in the park within the Swiftcurrent Creek drainage, downstream of Swiftcurrent Falls. Annual operation of the dam completely dewatered Swiftcurrent Creek downstream of the dam in the winter months, resulting in the death of native fish including bull trout (Mogen and Kaeding 2001). The associated St. Mary Irrigation Canal, used to deliver irrigation water to the Milk River, remains unscreened and results in the permanent loss of hundreds of bull trout and thousands of other native fish from the system each year (J. Mogen, USFWS, personal communication). The St. Mary Diversion Dam, used to provide water into the irrigation canal, creates an approximately 6' high impediment to upstream migration of bull trout during the migration season (Mogen and Kaeding 2005). Addressing the fishery impacts of this project would significantly improve migration conditions as well as survival of migratory bull trout.

On the west side of GNP, both migratory stocks of bull trout from Flathead Lake as well as populations that reside entirely within the park (known locally as "disjunct" migratory populations) are monitored. Flathead Lake migratory bull trout stocks underwent dramatic declines starting in about 1990, and declines are believed to have been the result of the introduction of Mysis shrimp *Mysis diluviana* into the system, resulting in major alterations in trophic dynamics (i.e. rapidly expanding lake trout population) in the lake, as well as drought conditions (Weaver et al. 2006). One of the most significant contemporary threats to these populations is predation with and competition by non-native fish species in both the migratory and rearing habitats of the Flathead River and Flathead Lake (Deleray et al. 1999, Muhlfeld et al. 2008).

The only redd counts that were completed in the Middle Fork Flathead drainage of the park were Harrison, Ole, and Park creeks (Table 1). However, the Ole and Park creek counts are biased low due to the poor counting conditions and should be viewed as absolute minimum counts. An early winter storm occurred in early October, dropping heavy snow and raising streamflow's across the park. The increased flow flattened the diagnostic "tails" of the bull trout redds making them very difficult to detect in much of the recently mobilized streambed.

Disjunct lake bull trout redd counts in the park's North Fork Flathead River drainage were limited in 2019. Only Quartz, Logging, and Akokala creeks were counted. Quartz Creek remained near the long-term average. No bull trout spawning activity was documented in Logging Lake and the lake is subject to a bull trout stocking and lake trout removal effort aimed at recovering this population. Only one redd was observed in Akokala Creek. Recent beaver activity has blocked access to Akokala Creek in the fall and as a result, bull trout now only have about 30 m of accessible spawning and rearing habitat. In addition, the limited habitat that is accessible has a high level of fine-sediment embeddedness that likely results in poor spawning and incubation success. The Akokala Lake bull trout population appears close to extirpation (Table 2).

High annual variability in counts can make detecting trends using redd counts difficult and require long data sets. Previous authors using similar data sets predicted it may take over 100 years of continuous redd count data collection before a statistically significant trend can be detected in some systems (Rieman and Myers 1997). However, evaluation of observer error in bull trout redd counts (Dunham et al. 2001, Muhlfeld et al. 2006), as well as documented relationships between redd counts and actual adult spawning escapement (Bonar et al. 1997, Dunham et al. 2001, Downs and Jakubowski 2006) support their continued use as a key monitoring tool for bull trout populations in GNP.

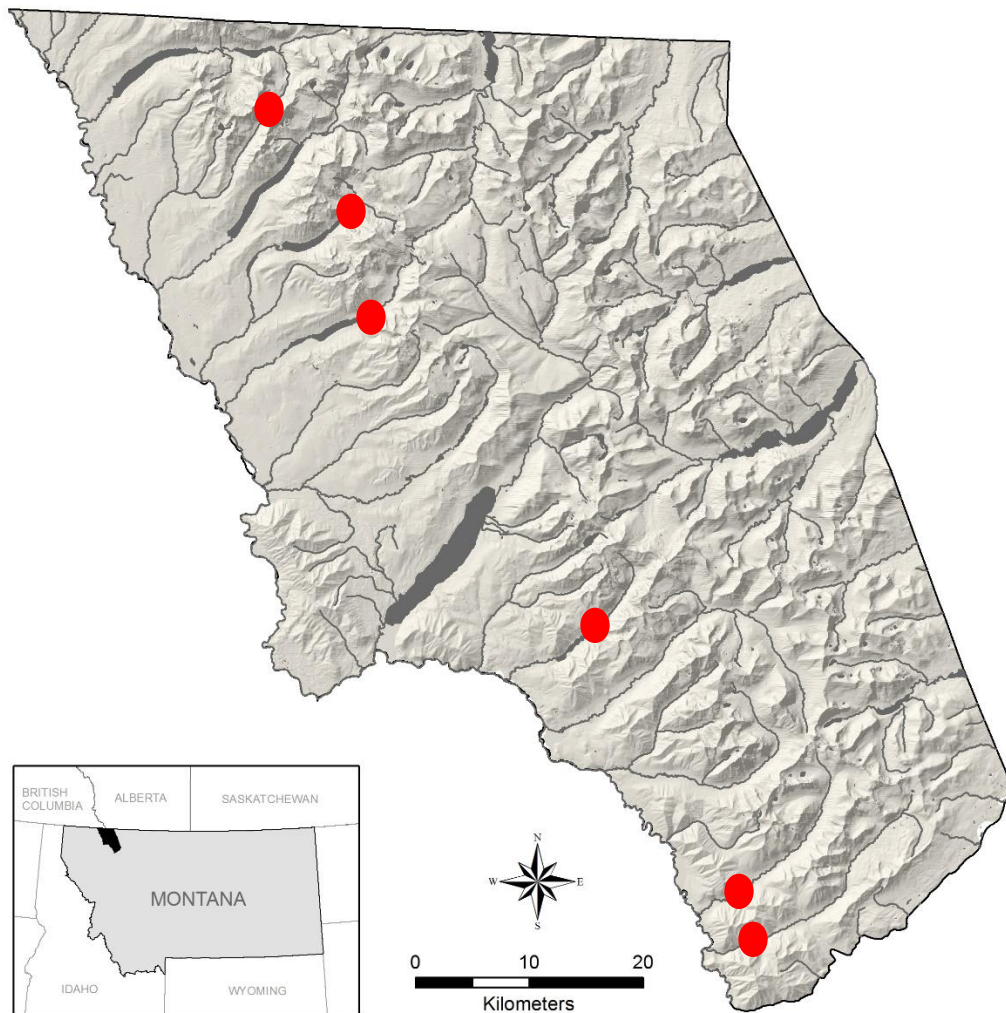


Figure 1. Drainages monitored for bull trout spawning activity (red circles) in Glacier National Park, Montana in 2019. They include Akokala, Quartz and Logging from north to south in the North Fork drainage and Harrison, Park, and Ole creeks from north to south in the Middle Fork drainage.

Expanding populations of lake trout have colonized almost all of the accessible lake habitats on the west side of GNP, and now threaten the persistence of the majority of the “disjunct” migratory bull trout populations remaining on the west side of GNP. Nine of seventeen lake-dwelling populations of bull trout located on the west side of GNP have been compromised by lake trout (Fredenberg et al. 2007), and lake trout have been documented replacing bull trout as the dominant predator in these waters, where long-term data on fish populations exists (Fredenberg 2002; Downs et al. 2011). Some populations appear to be persisting at dangerously low numbers (e.g. Bowman, Logging, and Harrison lakes), and interactions with non-native lake trout are likely the driving force behind the declines and the precarious status of bull trout in these systems (Donald and Alger 1993). In 2009, an experimental lake trout suppression program was initiated on Quartz Lake to preserve native fish populations, including bull trout, and is showing promise in reducing adult lake trout abundance. Similarly, a lake trout removal project was initiated on Logging Lake and was successful at removing large numbers of adult lake trout from 2015-2019. In addition, the NPS in partnership with both the US Fish and Wildlife Service and Montana Fish, Wildlife, and Parks have stocked 4,678 juvenile (age-0 and age-1) bull trout into Grace Lake and 2,152 age-1 bull trout into Logging Lake between 2016-2019. These bull trout came from wild eggs taken from Quartz Creek. Both of these projects are anticipated to continue through at least 2022.

The trend in bull trout redd counts in Quartz Creek remains stable despite a lower than average count of 27 (average=33) in 2019 (Figure 2). Adult bull trout were captured for spawn-take in upper Quartz Creek in 2016-2018. We will continue to monitor the population as the lake trout suppression project continues.

Table 1. Bull trout redd counts conducted on tributary streams to the Middle Fork of the Flathead River and the St. Mary River, Glacier National Park, 1980 to present.

Middle Fork Flathead Tributaries					St. Mary Tributaries			
	Ole Cr (g,p)	Park Cr (j,p)	Nyack Cr	Harrison Cr	Boulder Cr	Kennedy Cr (i)	Unnamed Kennedy Cr Trib.	Lee Cr
1980	19	--	14		--	--	--	--
1981	19	13	14		--	--	--	--
1982	51	0	23		--	--	--	--
1983	35	--	--		--	--	--	--
1984	26	--	--		--	--	--	--
1985	30	--	--		--	--	--	--
1986	36	87	27		--	--	--	--
1987	45	--	--		--	--	--	--
1988	59	--	--		--	--	--	--
1989	21	--	--		--	--	--	--
1990	20	--	--		--	--	--	--
1991	23	19	22		--	--	--	--
1992	16	1	12		--	--	--	--
1993	19	--	--		--	--	--	--
1994	6	--	--		--	--	--	--
1995	16	--	--		--	--	--	--
1996	10	--	--		--	--	--	--
1997	14	2	9		12	23	--	--
1998	22	--	--		42	37	--	--
1999	26	--	--		20	--	--	--
2000	33	10	13		30	23	--	--
2001	29	--	--		28	12	--	--
2002	21	--	--		28	11	--	--
2003	21	0	14		28	18	--	--
2004	14	--	--	4	27	13	--	--
2005	16	--	--	0	--	25	--	--
2006	31	--	--	8	50	20	--	--
2007	29	--	--	15	38	13	--	--
2008	42	23	16	14	58	22	--	--
2009	34	3	8	1	38	5	2	--
2010	32	3	6	6	33	12	0	--
2011	40	20	16	--	61	15	1	15
2012	53	15	14	1	53	9	--	31
2013	44	9	--	--	61	8	--	12
2014	32	4	11	10	66	20	--	16
2015	35	--	13	--	66	14	1	5
2016	31	--	--	--	--	--	--	22
2017	18	13	--	0	26	1	--	5
2018	20	11	14	--	39	0	--	7
2019	3	2	--	0	--	--	--	--

Table 2. Bull trout redd counts conducted in the North Fork of the Flathead River Drainage, Glacier National Park, 1994 to present.

	Akokala Cr (o)	Bowman Cr (c,l)	Logging Cr (n)	Lower Quartz	Middle Quartz (h)	Quartz Cr (a,b,e,m)	Rainbow Cr	Upper Kintla (m)
1994	--	--	--	--	--	--	--	52
1995	--	--	--	--	--	--	--	--
1996	--	--	--	--	--	--	--	--
1997	--	--	--	--	--	--	--	--
1998	--	--	--	--	--	--	--	--
1999	--	--	--	--	--	--	--	--
2000	--	--	--	--	--	--	--	--
2001	--	--	--	--	--	--	--	--
2002	--	0	--	--	--	--	--	--
2003	--	0	--	--	0	31	--	--
2004	--	0	3	1	0	46	--	--
2005	--	0	20	3	0	4	--	--
2006	--	2	0	2	0	36	--	--
2007	--	1	--	2	0	14	--	--
2008	11	0	5	3	0	51	28	0
2009	6	0	0	2	3	34	12	--
2010	1	1	3	2	0	27	4	25
2011	4	3	3	5	--	32	9	--
2012	5	--	1	3	--	33	--	--
2013	0	1	N/R	5	--	22	4	--
2014	12	4	0	4	1	66	5	--
2015	14	0	0	1	0	39	7	--
2016	--	--	--	--	--	--	--	--
2017	2	--	0	0	--	26	0	--
2018	4	--	0	6	--	35	--	--
2019	1	--	0	--	--	27	--	--

a=high flows may have obliterated some redds in 2005

b=spawning activity may have been impacted by weir placement near the mouth in 2007

c=minimum count during 2005 survey due to poor conditions

d=redd count accuracy may have been compromised by kokanee spawning activity in 2007 and 2008.

e=2008 data is a cumulative count based on multiple survey events

g=high flows may have obliterated some redds in 2004, 2005

h = old cutthroat redds still visible and may have affected count reliability

i = 3 additional redds were identified upstream of index reach in 2009

j=count was conducted by helicopter in 2008

l=poor counting conditions due to higher water levels and high turbidity (glacial) from Jefferson Creek in 2010

N/R=unreliable count due to kokanee spawning

m=2 redds observed in both 2008 and 2013, and 1 redd in 2014 and 2015 in Quartz above confluence of rainbow creek and included in index section total. This area (approx. 200m) not counted in other years.

n=2013 count likely inflated due to large (14-18") kokanee spawning at the time of the count. 14 redds not included that were presumed to be kokanee redds due to the observation of kokanee in the immediate vicinity of these redds.

o = New beaver dam complex located approx. 40 m from the lake blocking access to all upstream spawning habitat.

p = high flows made counting difficult and obliterated tails of redds. This count is biased low and should be viewed as a minimum count only and not for trend evaluation.

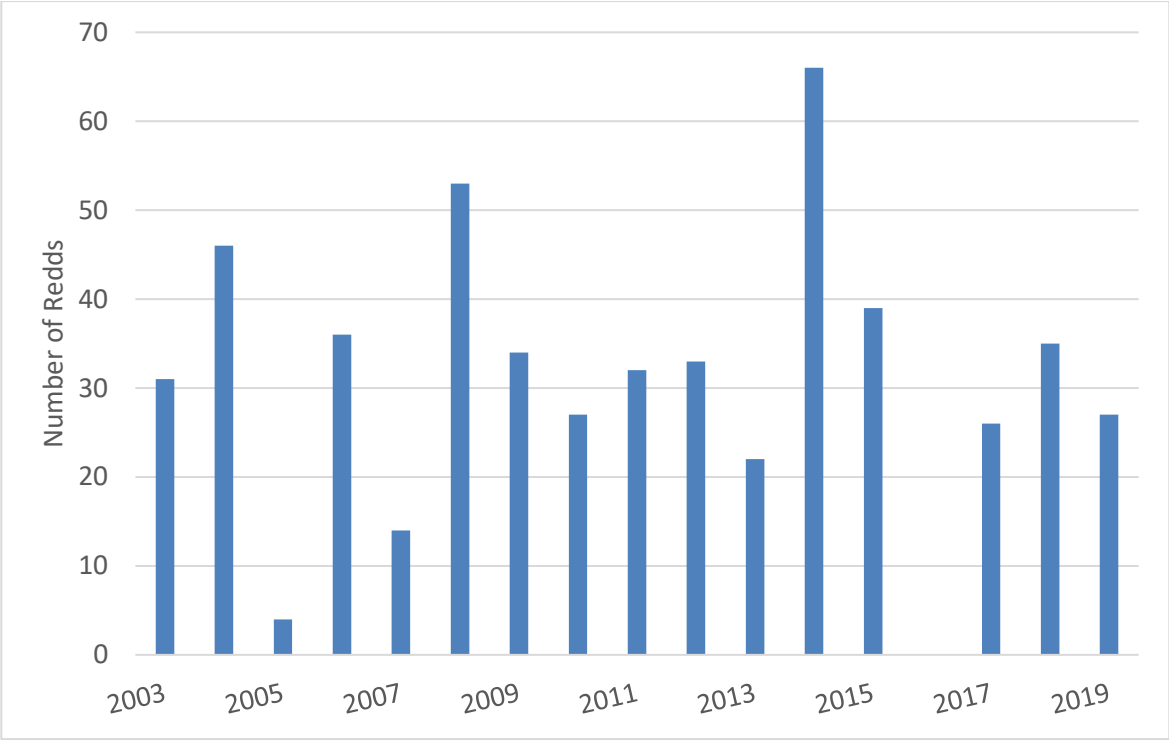


Figure 2. Bull trout redd counts in Quartz Creek, Glacier National Park, Montana.

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Aquatic Invasive Species Prevention and Monitoring

ABSTRACT

In 2019, Glacier National Park (GLAC) continued to implement a program of monitoring park waters for early detection of aquatic invasive species as well as mandatory boat inspection and launch permitting for all motorized and non-motorized watercraft using GLAC waters. Zebra or quagga mussels have not been found on any of the boats inspected since the inspection programs inception (nine years), although other aquatic species such as snails as well as aquatic plant fragments have been found on non-motorized watercraft. GLAC provides self-cleaning facilities (hoses and shop vacuums) for boaters to facilitate compliance with permit conditions. GLAC also provided financial support to the Confederated Salish and Kootenai Tribes as well as the Blackfoot Tribe to operate AIS stations on the reservations that also protect GLAC waters. Both tribes have intercepted mussel-infested boats in the past. Mussel detection dogs were also funded by the NPS and periodically deployed at park as well as neighboring tribal inspection stations. We continued the parks collaborative (through active participation in the state-led Upper Columbia Conservation Commission) AIS monitoring program, focused on water sampling for the presence of mussel larvae (veligers) and invasive mussel environmental DNA. Kintla Lake, Bowman Lake, Lake McDonald, Two Medicine Lake, St. Mary Lake, and Waterton Lake were monitored via plankton tows for veligers and environmental DNA in 2019. No AIS, aside from invasive fishes, have been detected in any park waters.

INTRODUCTION

Aquatic Invasive Species (AIS) are non-native species that negatively impact aquatic ecosystems, as well as human services and uses of those systems. AIS can impact native species and their habitats through a number of mechanisms including competition, predation, displacement, habitat disruption, or the spread of disease or parasites. Biological invasions by non-native species have become so widespread that they are significant contributors to global environmental change (Vitousek et al. 1996). For example, non-native invasive fish species have already had significant negative impacts on native fish populations within Glacier National Park (GNP) (Fredenberg 2002).

AIS such as zebra *Dreissena polymorpha* and quagga mussels *D. bugensis* present a growing worldwide problem. Native ecosystems rarely have established control mechanisms for such newcomers, and as such, they often establish at the cost of native flora and fauna. Impacts from aquatic invasive species can be extreme and affect ecosystems, recreation, and economics. AIS infestations are generally permanent; prevention is the best strategy to combat them. Public education is critical because many groups of aquatic invasive species need humans to move upstream.

Likely first introduced into the Great Lakes via trans-ocean ballast water transfer in 1986, zebra mussels were subsequently discovered in Lake St. Claire in 1988 (Griffiths et al. 1991). Zebra mussels have had a dramatic impact on aquatic ecosystems as well as public use of those ecosystems. Native to southern Russia, zebra mussels are efficient filter feeders and have the potential to reduce productivity of other aquatic species at higher trophic levels through lower trophic level competition for primary production (Ludyanskiy et al. 1993). For example, adult invasive mussels can filter up to 1L of water each day each and could strip out limited plankton resources and interrupt the native energy transfer pathways (e.g. plankton-zooplankton-fish-terrestrial/avian predators) in park watersheds. The quagga mussel, native to Ukraine, was not discovered in the Great Lakes until 1989 (Mills et al. 1996). Quagga mussels have been found to occupy deeper, colder areas in the Great Lakes than observed in their native range (as deep as 110m), broadening the potential impact area of these species from littoral to profundal areas of lakes (Mills et al. 1996). Quagga mussels have established populations in the large lower Colorado River basin reservoirs including Lake Mead. They are either male or female and use external fertilization to produce young (initially planktonic veligers), with an individual female capable of producing up to one million eggs/year. They spawn in the warmer summer months in north-temperate latitudes. Zebra mussels will initiate spawning at temperatures as low as 12°C, while quagga mussels have been documented spawning in colder waters (as low as 5°C) (Cohen 2007). The offspring are initially planktonic and spend several weeks floating in the water column before sinking to the bottom and colonizing the waterbody substrate as juvenile and adult mussels. The biological differences between zebra and quagga mussels suggest quagga mussels may be an even greater threat than zebra mussels.

Invasive mussels can also adversely impact native bivalve species through competition or by colonizing them as a host substrate and smothering them (Ricciardi et al. 1998). GLAC has some native bivalve species including fingernail clams and competition and displacement of these native bivalves could occur if invasive mussels became established in park waters. The only native mussel found west of the Continental Divide in Montana is the Western Pearlshell (Montana Field Guide; <http://fieldguide.mt.gov/displaySpecies.aspx?family=Margaritiferidae>; accessed 5/10/2017). It has not been identified as inhabiting the upper reaches of the Flathead drainage including the park, but its host fish, the westslope cutthroat trout, is abundant and widely distributed. Western Pearlshell mussels are found in the lower reaches of the Flathead River drainage and could be adversely impacted if invasive mussels colonized the drainage.

Aside from biological considerations, economic cost associated with management of invasive mussels is significant. It has been estimated that between 1989 and 2004, power generating and water treatment facilities in North America incurred approximately \$267 million in total economic costs dealing with zebra mussels (Connelly et al. 2007). Zebra and quagga mussels have continued to move south and west from their initial introductions and threaten to compromise native aquatic ecosystems across the west. Invasive mussels have been found on trailered watercraft in Montana, Idaho, and Washington (<http://nas.er.usgs.gov/taxgroup/mollusks/zebramussel/>) Invasive mussel veligers were recently discovered in Tiber Reservoir in Montana, less than 100 miles from the park's eastern boundary. In

addition, “suspect” invasive mussel veligers were also found in Canyon Ferry Reservoir, an impoundment of the headwaters of the Missouri River in Montana. However, additional monitoring failed to detect any additional mussel larvae in Canyon Ferry Reservoir for three subsequent monitoring years resulting in the removal of the reservoirs “suspect” classification in late 2019. Currently, only Tiber Reservoir remains classified as mussel-positive in Montana. That may change in the next 2-3 years if continued monitoring fails to detect adults or veligers in Tiber for five consecutive years.

Other AIS threaten GNP as well. Plant species such as Eurasian watermilfoil *Myriophyllum spicatum*, purple loosestrife *Lythrum salicaria*, and others are present within a three-hour drive of the park, and New Zealand mudsnails are present in southwest Montana. Taken together, the potential transport and establishment of additional AIS into park waters is a serious threat. In response, park managers are taking proactive steps to reduce the risk.

In 2009, GNP initiated a project to evaluate the risk of introduction and establishment of AIS in park waters (Downs et al. 2011) which documented the risk of potential AIS introduction and establishment. This evaluation, along with heightened awareness of the ecological, financial, and social impacts that AIS such as zebra and quagga mussels cause, prompted GNP to begin a limited boat inspection and launch permitting program in 2010. The initial program required a permit to launch motorized watercraft in park waters. In order to qualify for a launch permit, non-resident boaters were required to submit to an inspection of their boat for the presence of AIS. Because invasive mussels were not present in Montana at the time, an AIS-free self-certification program for resident motorized watercraft was implemented (rather than requiring NPS inspection). Resident boaters qualified for the launch permit by completing the AIS free self-certification form. However, due to the expanding nature of the AIS threat, including live mussels found on boats in Montana and invasive aquatic plants found in Flathead county, in 2012 the park expanded the program to require inspection and permitting of all non-hand propelled watercraft (regardless of Montana resident or non-resident status) before launch in any park water. The updated program included also an AIS-free self-certification requirement for all hand-powered non-motorized watercraft (e.g. canoes, kayaks) as they presumably present a lower risk of infestation and transport of AIS. This program continued in a similar fashion through the end of the 2016 visitor field season when the park closed its water to boating following the discovery of invasive mussels in Montana waters.

GLAC has also developed the *Glacier National Park Aquatic Invasive Species Action Plan*, which was authorized for implementation in 2014. This plan addresses GLAC-specific prevention, monitoring, education and outreach, and emergency response. It is intended to complement earlier NPS guidance on the issue of invasive mussels (NPS 2007). In accordance with the action plan, positive identification of invasive mussels in Montana led to the closing of all waters within Glacier National Park to motorized boating with the exception of concessionaire boats and GNP staff boats that remain in park waters at all times. After which, an expansion of the plan was put in place to allow for motorized visitor boats on Lake McDonald but only after the boats had been cleaned, drained, dried and quarantined for a period of thirty days.

METHODS

Upon entry to the park, boaters are informed by signage and/or entrance gate staff that a boat inspection and launch permit is required for all watercraft (i.e. motorboats, kayaks, canoes, rafts, etc.) launching within the park. Motorized boats wishing to launch on park lakes must be inspected, cleaned, drained, dried and quarantined for thirty days before a launch permit is issued. The launch permit remains valid as long as the boat does not leave the park. Another inspection and quarantine is required upon re-entry into the park if the boat was not sealed to the trailer by NPS staff when it departed the park. The park accepts sealed boats with proper documentation from State or Tribal partner inspection stations.

Boaters intending to boat on Bowman Lake are encouraged to have their boats inspected and sealed in West Glacier because it is difficult to keep a boat clean for inspection at Bowman Lake under wet road conditions on the North Fork Road, as well as staffing limitations. In the West Glacier vicinity, the busiest boat launch area (Lake McDonald),

inspectors are stationed near the boat launch. Waterton Lakes NP in Canada controls access to Waterton Lake, which spans the US-Canadian border. Waterton Lakes NP does not operate an identical inspection and permitting program but does inspect all non-motorized boats but does not permit launching of any motorized boats.

A public education effort regarding the risks of AIS to GLAC waters continued in 2019. The public has access to information on AIS produced by the Crown of the Continent Learning Center at GNP intended for the public addressing AIS threats to GNP (<http://www.nps.gov/glac/naturescience/ccrlc.htm>). The bulletin is brief (two pages) and is intended to provide key information regarding the status of AIS in GLAC, as well as how the public can help protect the park from additional AIS. GNP continued to provide AIS prevention content on its website (<http://www.nps.gov/glac/planyourvisit/outdooractivities.htm>) so boaters and other visitors would be exposed to the “clean, drain, dry” message before visiting the park. The AIS message was also incorporated into the Waterton-Glacier Guide brochure, available to the public.

We used plankton sampling equipment to monitor for the presence of invasive mussels in Bowman Lake (BW), Lake McDonald (MC), Two Medicine Lake (TWM), St. Mary Lake (SM), and Waterton Lakes (WAT)) (Table 1). In addition, electronic temperature recorders were installed in some of these same lakes near boat launches to provide information on the timing and likely duration of potential mussel spawning. This information informs the timing of our annual veliger sampling efforts.

Sampling for the presence of the larval form of the invasive mussels (i.e. veligers) was conducted according to established MFWP protocols (E. Ryce, MFWP, personal communication). Sampling occurred during peak surface water temperatures, after surface water temperatures had reached at least 54°F (12°C). This typically occurs in late July or early August. We used 30cm x 120cm x 64 micron x 9cm cod-end plankton nets to conduct vertical plankton hauls in each waterbody. The net was lowered into the water to a maximum depth of 20m or to within 1m of the lake bottom (if sample site depth < 20m) and then slowly retrieved to the water surface. We made triplicate hauls at each sample location and combined the contents of the three hauls into one 250mL plastic sample bottle resulting in one sample per site. Between 2 and 5 sites were sampled on each waterbody. In order to ensure we maintained the desired 67% alcohol content for preserving the samples, we filled the bottle no more than 1/3 full of sample and the remainder of the way with 200 proof non-denatured alcohol. The samples were shipped to the Montana Fish, Wildlife, and Parks AIS lab in Helena for processing. Veliger monitoring sites typically included a sample at the foot of the lake near the boat launch, one at mid-lake, and one at the upper end of the lake. GIS coordinates were taken for all sampling locations (Table 1).

Environmental DNA (eDNA) samples were collected using similar methods, but sampling is conducted in May and June of each year. Samples are processed in the field by spinning the sample collection cup to remove as much water as possible from the sample. The goal is to preserve the remaining organic matter in the sample in 100% non-denatured ethanol. A higher level of alcohol purity is required to preserve the samples for the eDNA analysis. eDNA samples were analyzed for the presence of invasive mussel DNA, and two invasive aquatic plants (Eurasian watermilfoil and curly-leaf pondweed) at the Montana Conservation Genetics Lab at University of Montana-Missoula.

Waterton-Glacier International Peace Park

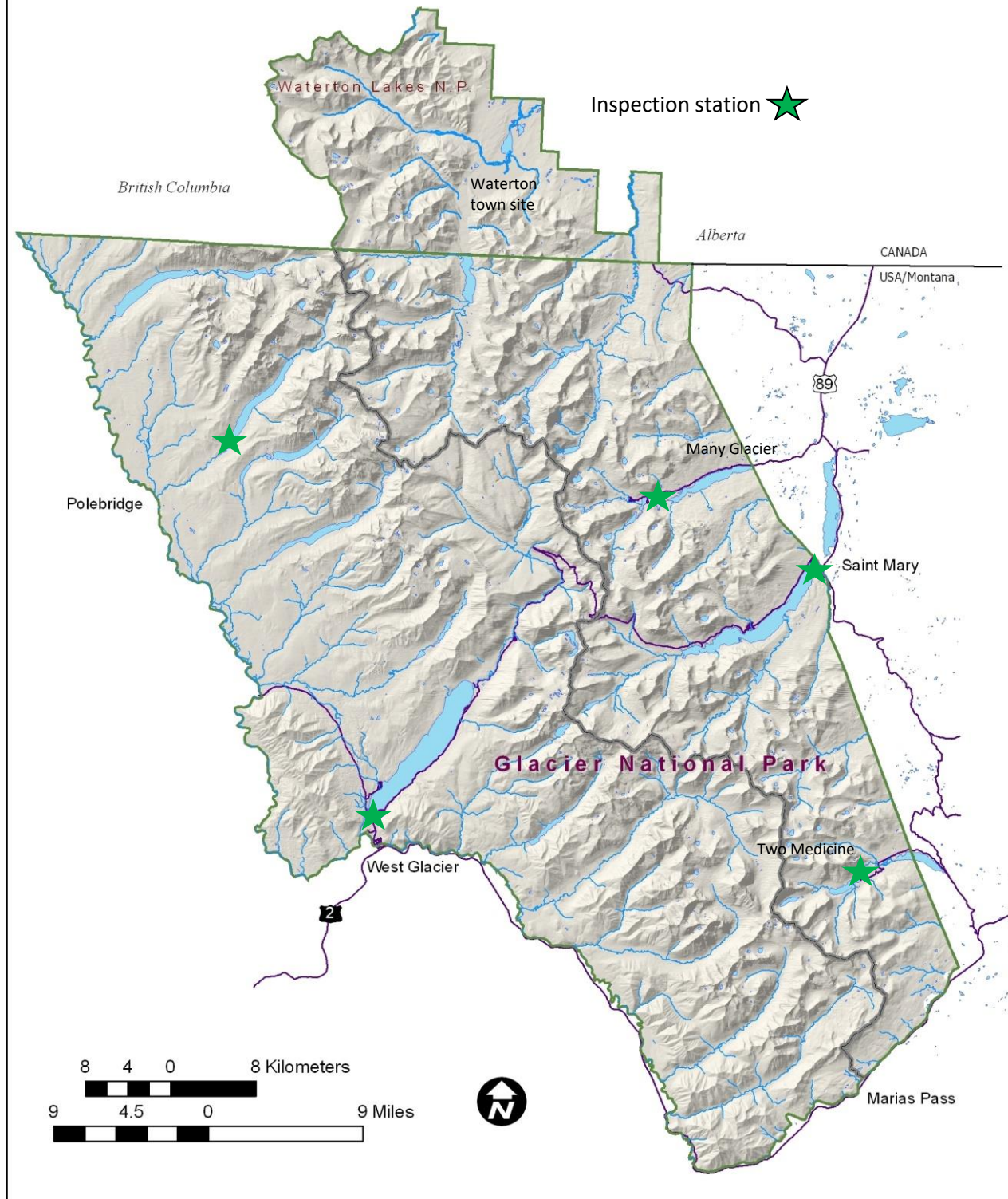


Figure 1. AIS boat inspection and launch permitting locations in Glacier National Park, 2019.

Table 1. Sample site descriptions for GLAC AIS sampling in 2019.

<u>Date</u>	<u>Sample type</u>	<u>water</u>	<u>Site Id.</u>	<u>Surface Water Temp.</u> (°C)	<u>Secchi Depth</u> (ft)	<u>Zone</u>	<u>UTM X</u>	<u>UTM Y</u>	<u>Notes</u>
6/5/2019	eDNA	Bowman Lake	BW 1	7.5	-	11	705896	5412597	Single haul
6/5/2019	eDNA	Bowman Lake	BW 2	7	-	11	705500	5412409	Single haul
6/5/2019	eDNA	Bowman Lake	BW 3	7	-	11	705397	5412481	Single haul
6/5/2019	eDNA	Bowman Lake	BW 4	8.5	-	11	707768	5416289	Single haul
6/5/2019	eDNA	Bowman Lake	BW 5	8	-	11	705743	5412151	Single haul
8/6/2019	Veligers	Bowman Lake	BW 1	16.9	41	11	705896	5412597	triple haul
8/6/2019	Veligers	Bowman Lake	BW 2	17.5	45	11	705500	5412409	triple haul
8/6/2019	Veligers	Bowman Lake	BW 3	17.7	44	11	705397	5412481	triple haul
8/6/2019	Veligers	Bowman Lake	BW 4	17.4	38	11	707768	5416289	triple haul
8/6/2019	Veligers	Bowman Lake	BW 5	18.9	-	11	705743	5412151	triple haul
6/6/2019	eDNA	Kintla Lake	KT 1	9	-	11	694476	5423762	Single haul
6/6/2019	eDNA	Kintla Lake	KT 2	8.5	-	11	694643	5423739	Single haul
6/6/2019	eDNA	Kintla Lake	KT 3	9	-	11	694586	5423659	Single haul
6/6/2019	eDNA	Kintla Lake	KT 4	8	-	11	694405	5423628	Single haul
8/6/2019	Veligers	Kintla Lake	KT 1	17.7	43	11	694476	5423762	triple haul
8/6/2019	Veligers	Kintla Lake	KT 2	17.5	57	11	694643	5423739	triple haul
8/6/2019	Veligers	Kintla Lake	KT 3	17.5	35	11	694586	5423659	triple haul
8/6/2019	Veligers	Kintla Lake	KT 4	18.3	-	11	694405	5423628	triple haul

6/14/2019	eDNA	Lake McDonald	MC 1	14	-	12	279246	5379419	Single haul
6/14/2019	eDNA	Lake McDonald	MC 2	13	-	12	286109	5389646	Single haul
6/14/2019	eDNA	Lake McDonald	MC 3	13	-	12	279054	5379533	Single haul
6/14/2019	eDNA	Lake McDonald	MC 4	15	-	12	287973	5389188	Single haul
6/14/2019	eDNA	Lake McDonald	MC 5	15	-	12	282478	5382948	Single haul
8/5/2019	Veligers	Lake McDonald	MC 1	18.6	33	12	279246	5379419	triple haul
8/5/2019	Veligers	Lake McDonald	MC 2	19.9	38	12	286109	5389646	triple haul
8/5/2019	Veligers	Lake McDonald	MC 3	18.6	30	12	279054	5379533	triple haul
8/5/2019	Veligers	Lake McDonald	MC 4	21.3	43	12	287973	5389188	triple haul
8/5/2019	Veligers	Lake McDonald	MC 5	19.9	19	12	282478	5382948	triple haul
5/30/2019	eDNA	Lake Sherburne	SHE 1	10.5	6	12	314817	5411429	Single haul
5/30/2019	eDNA	Lake Sherburne	SHE 2	10	8	12	314631	5411372	Single haul
5/30/2019	eDNA	Lake Sherburne	SHE 3	9	10	12	310726	5410503	Single haul
5/30/2019	eDNA	Lake Sherburne	SHE 4	9.5	9	12	312889	5411117	Single haul
5/30/2019	eDNA	Lake Sherburne	SHE 5	10	-	12	315034	5411449	Single haul
8/7/2019	Veligers	Lake Sherburne	SHE 1	18	12	12	314817	5411429	triple haul
8/7/2019	Veligers	Lake Sherburne	SHE 2	18	11	12	314631	5411372	triple haul
8/7/2019	Veligers	Lake Sherburne	SHE 3	17	14	12	310726	5410503	triple haul
8/7/2019	Veligers	Lake Sherburne	SHE 4	17	10	12	312889	5411117	triple haul
8/7/2019	Veligers	Lake Sherburne	SHE 5	17	-	12	315034	5411449	triple haul
5/28/2019	eDNA	Middle Waterton Lake	WAT 6	5.5	-	12	288034	5438579	Single haul

8/8/2019	Veligers	Middle Waterton Lake	WAT 6	17.2	-	12	288034	5438579	triple haul
6/3/2019	eDNA	St. Mary Lake	SM 5	8	-	12	320358	5402476	Single haul
6/4/2019	eDNA	St. Mary Lake	SM 1	7	-	12	314133	5396082	triple haul
6/4/2019	eDNA	St. Mary Lake	SM 2	7	-	12	314278	5396090	triple haul
6/4/2019	eDNA	St. Mary Lake	SM 3	8	-	12	320008	5401233	triple haul
6/4/2019	eDNA	St. Mary Lake	SM 4	8	-	12	319682	5401708	Single haul
8/14/2019	Veligers	St. Mary Lake	SM 1	14.8	27	12	314133	5396082	triple haul
8/14/2019	Veligers	St. Mary Lake	SM 2	14.8	27	12	314278	5396090	triple haul
8/14/2019	Veligers	St. Mary Lake	SM 3	15.5	37	12	320008	5401233	triple haul
8/14/2019	Veligers	St. Mary Lake	SM 4	15.4	21	21	319682	5401708	triple haul
8/14/2019	Veligers	St. Mary Lake	SM 5	15.5		21	320358	5402476	triple haul
6/12/2019	eDNA	Two Medicine Lake	TWM 1	8	-	12	324816	5372720	Single haul
6/12/2019	eDNA	Two Medicine Lake	TWM 2	8	-	12	324819	5372910	Single haul
6/12/2019	eDNA	Two Medicine Lake	TWM 3	8	-	12	324849	5373164	Single haul
6/12/2019	eDNA	Two Medicine Lake	TWM 4	8	-	12	325067	5373324	Single haul
8/7/2019	Veligers	Two Medicine Lake	TWM 1	16	24	12	324816	5372720	triple haul
8/7/2019	Veligers	Two Medicine Lake	TWM 2	16.2	39	12	324819	5372910	triple haul
8/7/2019	Veligers	Two Medicine Lake	TWM 3	16.2	30	12	324849	5373164	triple haul
8/7/2019	Veligers	Two Medicine Lake	TWM 4	16.4	-	12	325067	5373324	triple haul
5/28/2019	eDNA	Waterton Lake	WAT 1	5.5	-	12	288334	5426879	Single haul

5/28/2019	eDNA	Waterton Lake	WAT 2	5.5	-	12	288479	5427078	Single haul
5/28/2019	eDNA	Waterton Lake	WAT 3	5.5	-	12	287529	5431368	Single haul
5/28/2019	eDNA	Waterton Lake	WAT 4	5.5	-	12	287794	5437616	Single haul
5/28/2019	eDNA	Waterton Lake	WAT 5	5.5	-	12	287627	5437649	Single haul
8/8/2019	Veligers	Waterton Lake	WAT 1	15.3	25	12	288334	5426879	triple haul
8/8/2019	Veligers	Waterton Lake	WAT 2	15.2	29	12	288479	5427078	triple haul
8/8/2019	Veligers	Waterton Lake	WAT 3	15.2	28	12	287529	5431368	triple haul
8/8/2019	Veligers	Waterton Lake	WAT 4	16.5	25	12	287794	5437616	triple haul
8/8/2019	Veligers	Waterton Lake	WAT 5	15.5	28	12	287627	5437649	triple haul

RESULTS AND DISCUSSION

Water temperatures were recorded throughout the high boating-use summer months on waters with developed boat ramps. Zebra mussels will spawn when water temperatures reach and remain above 12°C (54°F) while quagga mussels may spawn at even cooler temperatures (down to 5 °C (41 °F) (Cohen 2007). However, preferred or optimal spawning temperatures for both species are at or above 20 °C. All park waters sampled had extended time-periods when water temperatures remained consistently above the 12°C (54°F) threshold level in 2018 and 2019 (Figures 2-5). For example, St. Mary Lake which generally has the coldest summer surface temperatures of the lakes monitored, remained above 12 °C for approximately a month in each of the summers of 2018 and 2019. Water temperature measurements suggest a potential summer spawning season for invasive mussels in park lakes lasting from 1 to 2+ months. Although calcium levels haven't been measured in lakes across the park for several decades, it is unlikely they have changed appreciably. Previous evaluations (Downs et al. 2011) suggest sufficient calcium (>15mg/L Ca) in the park lakes monitored in this study to support mussel shell growth, with Two-Medicine Lake being the only potential exception (<10mg/L Ca).

We timed our invasive mussel veliger sampling (plankton hauls) to correspond with peak water temperatures to increase the probability of detection. We sampled the most likely areas to become infested early in an invasion (i.e. boat ramp and dock areas) within each water sampled. We anticipate conducting veliger sampling again in 2020 as well as the eDNA sampling effort with the Flathead Lake Biological Station to continue to complement the veliger sampling. eDNA sampling offers some potential advantages, the most beneficial perhaps is the ability to sample the water for invasive mussels at any time of the year. Veliger sampling is limited to the fairly short warm-water sampling window of late summer when mussels would be expected to be reproducing. By using both approaches we improve the likelihood of detection the presence of invasive mussels early in an infestation, potentially affording the opportunity for earlier emergency response activities.

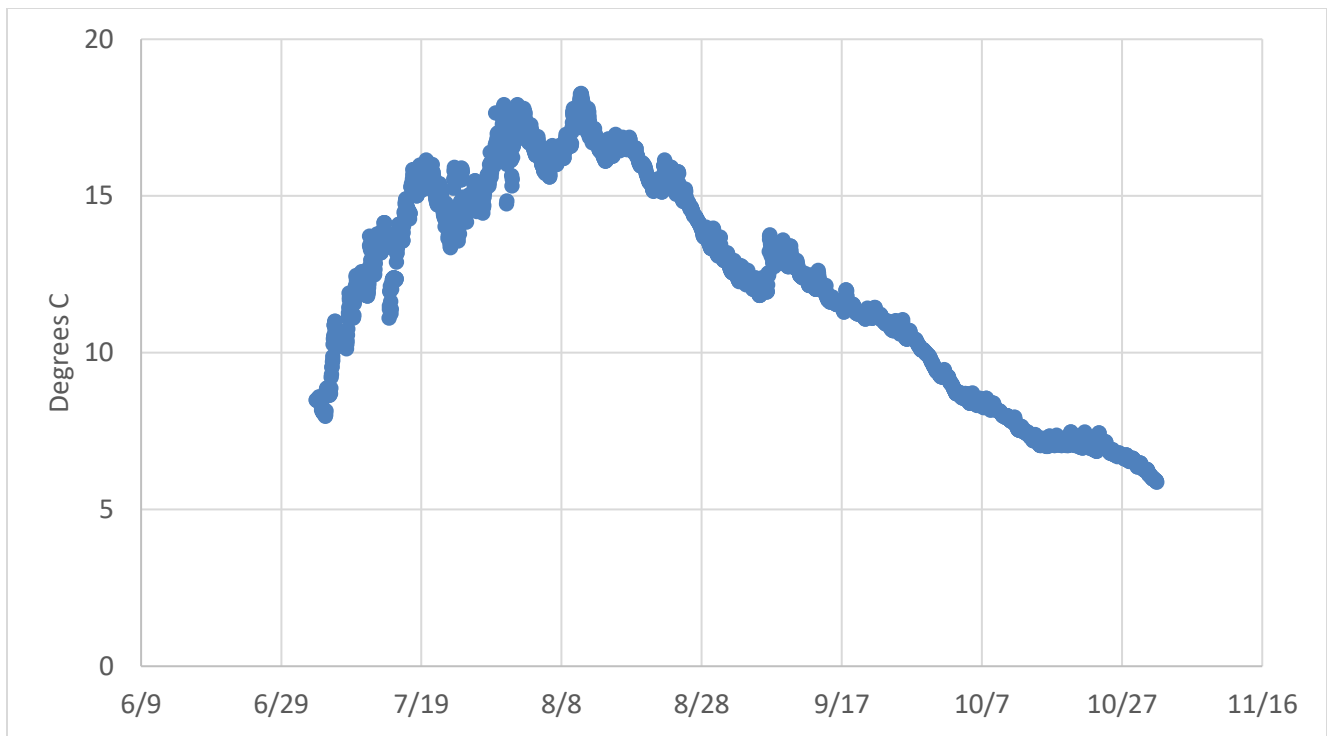


Figure 2. Two-Medicine Lake water temperatures recorded near the lake bottom at the end of the NPS boat dock in 2018.

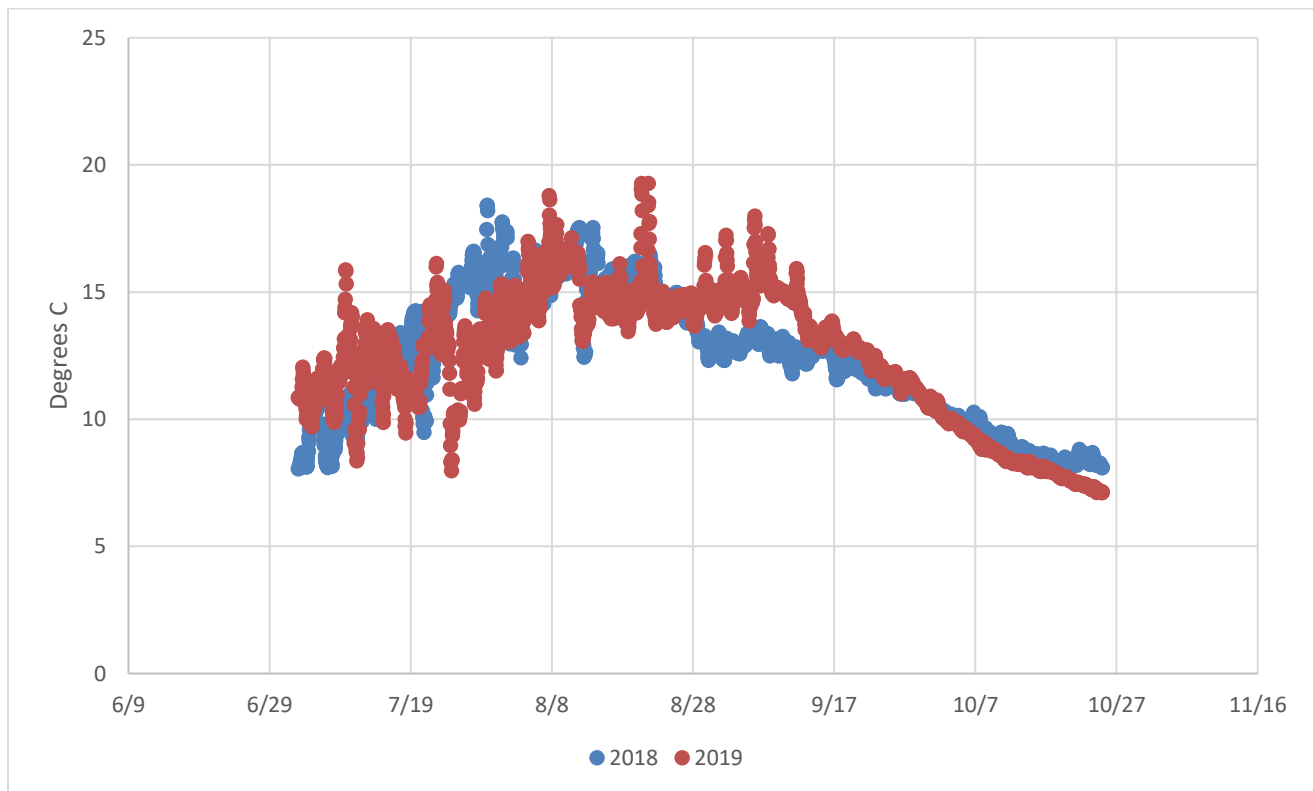


Figure 3. Water temperatures recorded near the bottom at the end of the boat dock on St. Mary Lake in Glacier National Park in 2018 and 2019.

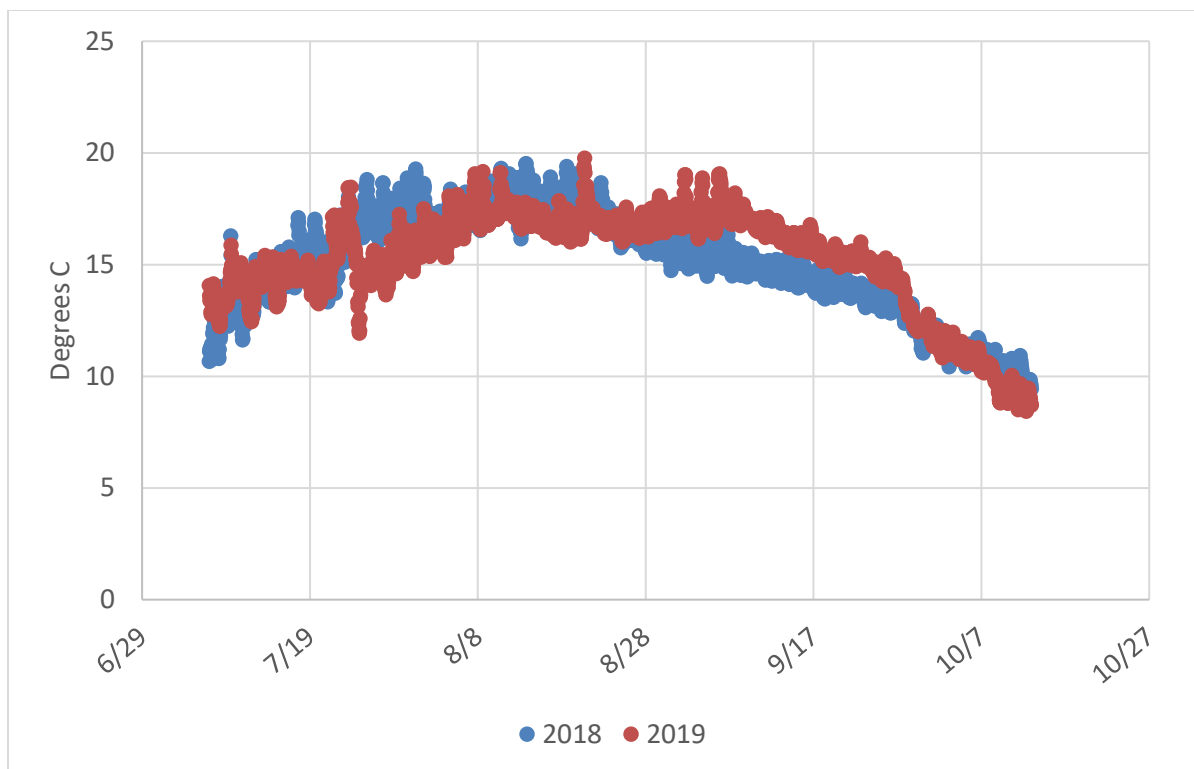


Figure 4. Bowman Lake water temperatures recorded near the lake bottom at the end of the NPS boat dock in 2018 and 2019.

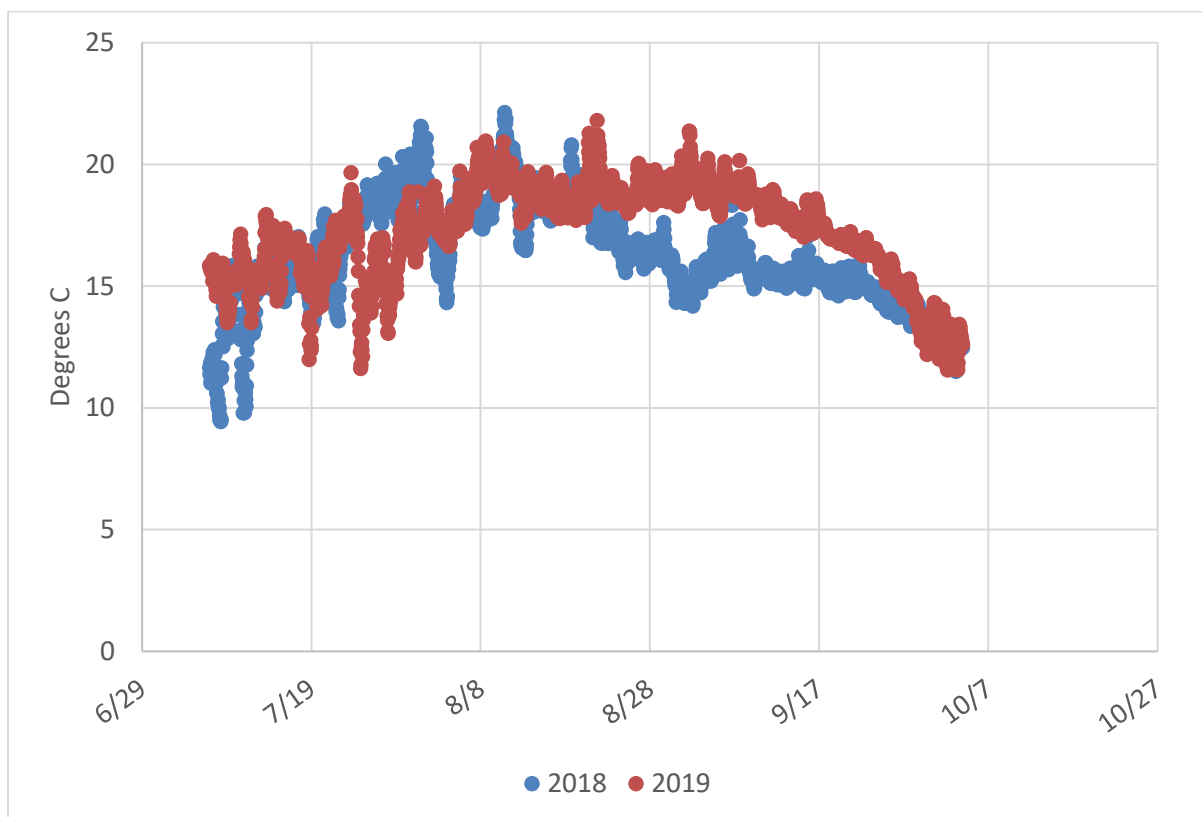


Figure 5. Lake McDonald Lake water temperatures recorded near the lake bottom at the end of the NPS boat dock in 2018 and 2019.

Glacier's AIS prevention program began in 2010 and has steadily improved over time. The detection of invasive mussel larvae in the nearby Tiber and Canyon Ferry Reservoirs in Montana in the late summer/fall of 2016 prompted implementation of the park's AIS Emergency Response Plan starting in November 2016. All lakes were immediately closed to all boating activity and motorized boat launch points to lakes were gated or otherwise closed. All waters in Glacier remained closed to motorized and/or trailer launched boats until early July when Lake McDonald was opened to permitted motorized boats that have been cleaned, drained and dried and quarantined for at least 30 days prior to launching. This standard has been implemented on other park lakes in recent years, with motorized boating currently being allowed on most-park waters where it was allowed on prior to 2016 (subject to a 30-day quarantine). In addition, the park began inspecting and permitting all non-motorized boats (including kayaks, canoes, rafts, etc.). This has resulted in inspection numbers exceeding 10,000 watercraft in recent years. To obtain these permits, all watercraft must be clean, drained and dried and pass inspection at one of the park inspection stations. According to Montana guidelines, Canyon Ferry Reservoir was removed from the "suspect positive" mussel category since evidence of their continued presence has not been observed for the past three years. Tiber Reservoir remains listed as a mussel positive water, but it too is in the process of working towards delisting because there have been no detections of mussel larvae using either veliger sampling or eDNA for the past two years. As new AIS issues and threats arise or current ones change, park managers will continue to be challenged in finding a balance between accommodating visitor use of GNP and providing vigilant ecosystem protection and conservation.

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